Approach for Estimating Changes in Blood Lead Levels from Lead Wheel Weights

Peer Review Draft Report

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EXECUTIVE SUMMARY

EPA is evaluating options to address the potential risks posed by lead wheel weights. These wheel weights can be lost from cars and can enter the environment, leading to potential exposures to children and adults who inhale or ingest roadway particles containing wheel weight lead or who drink contaminated water. In addition, wheel weights may be collected by home hobbyists and melted for use in making bullets, fish sinkers, or other hobby items.

There is a large database of studies on the health effects associated with lead, focusing primarily on neurological, cardiovascular, immune, reproductive, and blood effects. However, there are also studies examining associations between lead exposure and effects on the hepatic system, gastrointestinal system, endocrine system, bone and teeth, ocular health, respiratory system, and cancer. Neurocognitive effects in children are of particular concern due to the increasingly lower levels at which they have been reported and the potential for lifelong impact. Recent studies have reported negative associations between blood lead concentrations in children and IQ, as well as neurocognitive effects such as reading and verbal skills, memory, learning, and visuospatial processing, at blood lead concentrations as low as 2 µg/dL. In addition, studies focusing on behavioral problems, such as anxiety, distractibility, conduct disorder and delinquent behavior, have noted effects at blood lead levels ranging from 3-11 µg/dL. These new studies, which have examined blood lead levels in the range of 0.8 to >10 µg/dL, strengthen the evidence that there may not be a threshold associated with blood lead exposures. Studies on other health outcomes have reported effects at blood lead concentrations between 5-10 µg/dL or higher.

This approach document investigates the exposure to lead wheel weights in two exposure scenarios. In the first case, wheel weights are lost from vehicles near the roadway and eroded due to abrasion with other vehicles or debris. The lead is then released to the air as part of the roadway dust due to turbulence from the wind field or from passing vehicles. As this lead migrates to nearby homes, it can enter the yard soil or the indoor dust. Children or adults living nearby can be exposed through inhalation of the contaminated air or ingestion of soil or dust particles.

To investigate this first exposure scenario, a series of modules were developed to estimate the 1) release of wheel weight lead from the roadway, 2) the dispersion and deposition of this lead in the air to nearby yards, 3) the associated soil concentration in the yard, 4) the indoor dust concentration due to track-in of contaminated yard soil, and 5) the associated blood lead for children and adults in a near-roadway home. Whenever possible, existing peer-reviewed models or equations were used as the estimation tools for each module. However, new analysis tools were created for this assessment to estimate the road dust emission, the soil concentration, and the dust concentration. For the road dust emission and soil concentration, the analysis tools are simple mass-balance equations. For the dust concentration, a regression relationship was developed based on house survey data to relate dust concentration to soil concentration and housing vintage.

The selection of all parameters and modeling techniques is thoroughly documented in the report.

The incremental increase in blood lead levels that result from lead wheel weights that degrade in the environment is the desired metric from the analysis, not absolute blood lead levels. The estimates of exposure to lead in wheel weights were calculated by subtracting blood lead levels when wheel weight exposure is zero from the total blood lead for five different exposure conditions. The conditions differ in terms of general location (urban, downtown rural, or suburban), housing vintage (either before 1940 or after 1980, with older homes having higher dust and soil background lead due to the presence of lead-containing paint), and soil concentration (either high or low background concentration). In each exposure condition, modeled homes were placed in different exposure categories according to the magnitude of the modeled air lead concentration.

For children aged 0 to 7, the changes in blood lead level from lead in wheel weights vary from under 0.01 to 0.25 μg/dL. For adults, the changes in blood lead level from lead in wheel weights vary from less than 0.01 to 0.07 μg/dL.

Several parameters, particularly those affecting the magnitude of lead released to the air from the roadway, are poorly described in the literature and are subject to large uncertainty. These include the wheel weight loss rate from vehicles, the wheel weight removal rate from the roadway, the wheel weight degradation rate, the roadway dust loss rate, the yard soil depth, and the yard soil lead residence time. Efforts have been made to select the most reasonable value for each parameter from those available. The effect of varying these parameters is examined in the uncertainty analysis. Changing each parameter one at a time to values giving lower blood lead levels (either higher or lower parameter values, depending on their use in the module equations) results in child blood lead levels that are two to five times lower than those reported in the main analysis.

The second exposure scenario captures high-end exposure for a home hobbyist who melts lead to make hobby items such as bullets or fish sinkers. Owing to the lack of specific descriptive data about these activities in the literature, air concentrations were estimated using a saturation vapor pressure equation. Floor lead dust loadings following the melting event were estimated using a simple mass balance model. The vapor pressure concentrations were estimated at two representative temperatures, 316^{o}C (600^{o}F) and 454^{o}C (850^{o}F). These temperatures resulted in air concentrations of 0.24 and 15.7 μg/m^3. The dust loadings from the melting event were 0.18 and 11.4 μg/ft^2.

In order to support the cost-benefit analysis for the lead wheel weights rule, IQ decrements due to the lead in wheel weights are estimated for the near-roadway scenario.

1. INTRODUCTION

EPA is evaluating options to address the potential risks posed by lead wheel weights. These wheel weights can be lost from cars and can enter the environment, leading to potential exposures to children and adults who inhale or ingest roadway particles containing wheel weight lead or who drink contaminated water. In addition, wheel weights may be collected by home hobbyists and melted for use in making bullets, fish sinkers, or other hobby items.

{Summary paragraph on lead hazard concerns under revision.}

This document describes an approach for estimating exposure concentrations and/or blood lead levels for two exposure scenarios. In the first, or "Near Roadway Scenario", an adult and child are considered to reside near a roadway in three case study locations: the residential portion of an urban environment, the downtown of a suburban environment, or the downtown of a rural environment. The general framework for the exposure assessment approach is shown in Figure 1. First, the exposure media concentrations are estimated. Lead is emitted from the roadway after the abrasion and pulverization of lead wheel weights, and the lead migrates to the yard, resulting in air lead concentrations and inhalation exposure. In addition, the lead deposits in the yard soil and migrates into the indoor environment as dust, resulting in oral exposure. These media exposures are modeled using a combination of peer-reviewed models and simple mass-balance techniques, as described in Sections 4.1 to 4.4. Media concentration results are provided in Section 4.6.

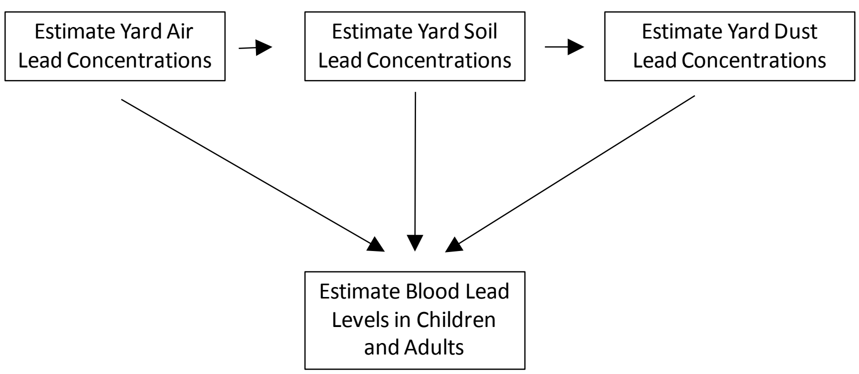

Figure 1. Flowchart Showing the Assessment Approach for the Near-Roadway Residence Exposure Scenario

After estimation of the media lead exposure concentrations, the model used to predict children's blood lead impacts was EPA's Integrated Exposure Uptake Biokinetic Model for Lead in Children (IEUBK) (USEPA 2010c). Because the IEUBK model can only be used up to an age of 84 months, the Adult Lead Methodology (ALM; U.S. EPA, 1996) was to estimate blood-lead impacts in adults. Section 4.5 provides details about the blood lead model implementation, while Sections 4.7 and 4.8 present the results.

In the second exposure scenario, or "Home Melting Scenario", a home hobbyist is assumed to melt lead from wheel weights in order to cast bullets, fishing weights, or other hobby items. The general framework for the exposure assessment approach is shown in Figure 2. The melting is assumed to occur inside a garage with the possibility of a child and adult present during the event. Air lead concentrations and garage dust loadings resulting from a single hour melting event are estimated using a saturation vapor pressure technique and a simple mass-balance technique, as described in Section 5. The Home Melting Scenario is characterized as a high end exposure estimate, which would fall in the upper end of the distribution of exposures. Many practices are used by home hobbyist to minimize exposure, the most significant being locating the melting pot outdoors. Due to the large number of permutations and combinations of exposure variables for the home melting scenario, a high end estimate is valuable for evaluation of the highest potential exposures for both children and adults to a hypothetical single hour-long event. Because of the uncertainties associated with the high-end estimate, blood lead levels were not estimated for this scenario. Finally, it should be noted that while hobbyists do use wheel weights as a potential source of lead for casting, other sources of lead are also available for hobbyist applications.

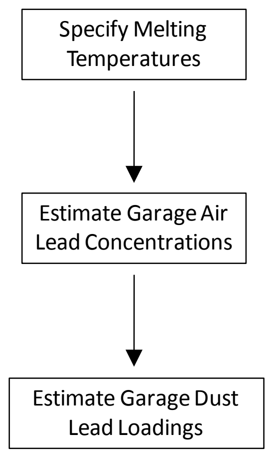

Figure 2. Flowchart Showing the Assessment Approach for the Home Melting Exposure Scenario

In order to facilitate an economic cost-benefit analysis in support of the proposed wheel weight rule, IQ decrements were selected as the health-endpoint for children. A piecewise-linear relationship from the Lanphear pooled analysis (Lanphear et al., 2005) was used to estimate IQ decrements from blood lead levels for the Near-Roadway Scenario only, as described in Section 6.

2. HEALTH HAZARD SUMMARY

This section is under revision and is not the subject of the Peer Review.

3. ENVIRONMENTAL FATE AND TRANSPORT

Lead wheel weights can be dislodged and then lost from vehicles, thus releasing lead into the environment. Root (2000) estimated that 1,650 tons (3.3 million pounds) per year of lead wheel weights are thrown from vehicle wheels and are deposited onto American streets in urban areas. The U.S. Geological Survey estimates that in 2003, 2,000 tons (4.0 million pounds) of lead wheel weights were lost on all of the nation's roads (USGS, 2006). When the wheel weights are lost from wheels they may fall onto road surfaces, where grinding and impacts between vehicle tires and road surfaces may break them into pieces or pulverize them into dust, to which exposure can occur. The amount of this breakage and pulverization will vary based on many factors including but not limited to: how far the lead wheel weight skids once it is thrown from the vehicle, the contact time with road surfaces during its travel when lost from the vehicle, whether it is hit by subsequent vehicles, and whether the lead wheel weight comes to rest on the median strips or curbs and is inaccessible for further abrasion/pulverization from vehicular traffic.

Lead sorbs strongly to soil constituents and is only weakly soluble in pore water; therefore it is essentially immobile in soil except under acidic conditions (ATSDR, 2007). The sorption of lead in soil is dependent on the pH, the organic matter content, the cation exchange capacity, the presence of inorganic colloids and iron oxides, soil type, particle size, and the amount of lead present. Lead is strongly chelated by humic or fulvic acids in the soil (ATSDR, 2007). In addition to sorption, lead can be immobilized by precipitation of insoluble salts such as carbonates, sulfates, sulfides and phosphates (HSDB, 2005). Most lead is retained strongly in soil, and very little is transported through runoff to surface water or leaching to groundwater. The solubility of lead in soil is dependent on pH, being sparingly soluble at pH 8 and becoming more soluble as the pH approaches 5. Between pH 5 and 3.3, large increases in lead solubility in soil are observed. These changes in lead solubility appear to correlate with the pH-dependent adsorption and dissolution of Fe-Mn oxyhydroxides (ATSDR, 2007). When released to soil, lead is expected to convert to more insoluble forms such as $PbSO_4$, $Pb_3(PO_4)_2$, PbS and PbO (HSDB, 2005). When metallic lead particles are released to soil, the lead surface reacts with air to form lead oxides. These lead oxides rapidly react with CO_2 from air or carbonates and sulfates from the soil to form a layer of cerussite ($PbCO_3$), hydrocerussite [$Pb_3(CO_3)_2(OH)_2$] and anglesite ($PbSO_4$) which appear on the surface as a white crust material. In the environment, these compounds form a protective surface coating that inhibits further corrosion of the metallic lead; however, cationic Pb^{2+} is eventually released (Vantelon et al, 2005, Lin et al., 1995).

When released to aquatic environments, a large fraction of lead introduced will be associated with suspended solids that settle down into the sediments. The amount of lead that can remain in solution in water is a function of the pH and the dissolved salt content. Equilibrium calculations show that the total solubility of lead in hard water (pH >5.4) and soft water (pH <5.4) is 30 µg/L and 500 µg/L respectively (U.S. EPA, 1977). At the low concentrations at which lead is normally found in the aquatic environment, most of the lead in the dissolved phase is complexed by organic compounds. The organic complexation increases with increasing pH and decreases with increasing water hardness (Callahan, 1979).

When released to the atmosphere, lead-bearing particles are transported to soil and water by wet deposition (rain and snow) and dry deposition (gravitational settling and deposition on water and soil surfaces). Approximately 40–70% of the deposition of lead is by wet deposition, and 20–60% of particulate lead once emitted from automobiles is deposited near the source. An important factor in determining the atmospheric transport of lead is particle size distribution. Large particles, particularly those with aerodynamic diameters of >2 μm, settle out of the atmosphere fairly rapidly and are deposited relatively close to emission sources (e.g., 25 m from the roadway for those size particles emitted in motor vehicle exhaust in the past); smaller particles may be transported thousands of kilometers (ATSDR, 2007). Lead particulates resulting from pulverized wheel weights are likely to exist as relatively large particles and are expected to deposit within a few meters of the roadside from which they were released. Such particles are not expected to possess the small aerodynamic diameters that would allow for long range transport.

4. NEAR-ROADWAY EXPOSURE SCENARIO

In the Near-Roadway Scenario, lead is released into the roadway environment due to degradation/pulverization of lost wheel weights, the lead migrates to the air surrounding the home, the deposition of lead particles contributes to yard soil concentrations, and indoor air and outdoor soil lead levels influence the indoor dust lead levels. Wheel weight lead may also contaminate groundwater. However, it was assumed that the exposed population obtained water from city reservoirs and the wheel weight lead contribution to drinking water was not included.

In order to estimate exposure to lead in this scenario, the fate and transport of lead from wheel weights in the air, soil, and indoor dust must be quantified. Next, the lead exposures in air, soil, and indoor dust can be combined with background exposure in all media to estimate the incremental effect of lead wheel weights on child and adult blood lead levels. A literature search was undertaken to determine what existing models and data could be used in the assessment. In general, data describing the physical process of lead wheel weight loss and degradation/pulverization on the roadway are sparse, making input parameters related to these processes uncertain. In keeping with the EPA exposure assessment guidelines where data is sparse, the analysis framework favors less data-intensive modeling techniques. Where possible, existing models used in other lead exposure assessments and suggested in EPA's Guidelines for Exposure Assessment were applied to this assessment. However, in cases where a suitable peer-reviewed model could not be found and applied, the scarcity of lead wheel weight data dictated the use of simple mass-balance models and empirical regression over the creation of more complicated models.

Figure 3 presents a flow chart that shows the inputs (free text), models (boxes), intermediate outputs (ovals) and final outputs (diamonds) used in the assessment framework. The gray boxes show the areas where simple mass-balance models and empirical regressions were adapted for this analysis, while the white boxes represent existing peer-reviewed models that were applied using protocols from other EPA lead exposure assessments. The overall framework is a system of connecting modules which estimates the necessary media concentrations (outdoor air, yard soil, and indoor air and dust) and the resulting blood lead (adults and children).

Estimates are first made for blood lead resulting from exposure to lead from all sources (both wheel weights and other sources) in all media (air, soil, dust, drinking water, and diet). This exposure is assumed to include the contribution from wheel weights and is called the "total" blood lead estimate. Then, the assessment modules are used to estimate the contribution to lead in air, soil, and dust from lead wheel weights alone. These media levels are used to find the hypothetical lead levels in the exposure media if the adult and child had not been exposed to lead from wheel weights by subtracting the wheel weights contribution from the total media lead level. These "no wheel weights" media levels are used to estimate the "no wheel weights" blood lead levels. Finally, the two blood lead estimates can be used to estimate the incremental change in blood lead due to the presence of lead from wheel weights by subtracting the “no wheel weights” blood lead estimates from the “total” blood lead estimates.

The assessment modules were applied to five case study locations: two urban scenarios, two rural scenarios, and one suburban scenario. These scenarios are not intended to be nationally representative. Instead, they are intended to capture exposures for five sets of hypothetical populations living near roadways. In each case, a proxy city was selected to aid in the development of input parameters for the assessment modules. The five case study scenarios are shown in Table 1. In each of the above scenarios, the particular proxy city was selected because it had the general characteristics of the target population locale and because there were existing data for some of the necessary parameters, such as traffic volume or soil concentration.

The urban scenario is intended to reflect the inner city section of a large metropolitan area with multi-unit homes and yard areas. As such, the soil concentrations are selected to be fairly high (see Section 4.3.2) to reflect the fact that many inner city locations have high soil concentrations due to the historic use of lead in gasoline and lead in paint. However, two different housing vintages (Scenarios A and B) were selected to reflect the fact that homes may be quite old (with higher background dust concentrations due to the presence of lead-containing paint) or may be of newer or refurbished construction (with lower background dust concentrations and no lead-containing paint). The city of Dorchester, MA was selected to serve as a proxy city for determining model parameters (see Sections 4.1.2, 4.2.2, and 4.3.2).

The rural scenario is intended to reflect the downtown area of a rural town. In such a town, homes may be of older or newer construction and may be built on areas of low or high historical soil lead contamination. For this reason, two scenarios (C and D) were constructed to represent a higher overall lead exposure (higher soil and older construction) and a lower overall lead exposure (lower soil and newer construction). The city of Boulder, MT was selected to serve as a proxy city for determining model parameters (see Sections 4.1.2, 4.2.2, and 4.3.2).

Finally, the suburban scenario is intended to reflect the downtown area of a suburban city. Most suburbs have relatively newer construction and lower overall historical soil contamination. Thus, only a single suburban scenario (E) was selected with lower soil and newer housing vintage. The city of Turners Falls, MA was selected to serve as a proxy city for determining model parameters (see Sections 4.1.2, 4.2.2, and 4.3.2).

Numerous inputs were needed for the different assessment modules. The data quality for each input varied according to the robustness of the data source and the degree of variability in the data. Variables shown in boldface type represent the most uncertain and/or variable parameters. In setting the input parameter values for the analysis, many parameters had a range of values in the literature. Parameters with a substantial amount of variation and uncertainty (such as the lead in soil residence time (Table 12)), were set at a value near the high end of its range in order to ensure that exposures in the hypothetical populations were not being underestimated. However, parameters that were well characterized in the literature or other exposure assessments, such as background dietary and water lead exposure concentrations, were selected as average or median values. Although each variable was selected based on the availability of quality data to yield a plausible value, no attempt was made to trace the probability of all parameters taking on the specific permutations and combination of values used in the assessment. However, the effect of choosing alternative values for the most uncertain and/or variable parameters which will yield lower wheel weight lead media concentrations is explored in the uncertainty analysis in Section 4.9.

Table 1. Assessment Scenarios in the Near Roadway Lead Wheel Weights Exposure Analysis

Scenario 1	Scenario 2	Scenario 3	Scenario 4	Scenario 5
Urban, high soil, pre-1940 housing vintage	Urban, high soil, post-1980 housing vintage	Rural, high soil, pre-1940 housing vintage	Rural, low soil, post-1980 housing vintage	Suburban (low soil, post-1980 housing vintage)

Sections 4.1 to 4.5 describe the approach adopted for each assessment module and details about the selection of input parameters. Section 4.6 presents the lead exposure concentrations in the air, soil, and dust media. Section 4.7 presents the blood lead levels for children and Section 4.8 presents the blood lead levels for adults. Finally, Section 4.9 presents the uncertainty analysis.

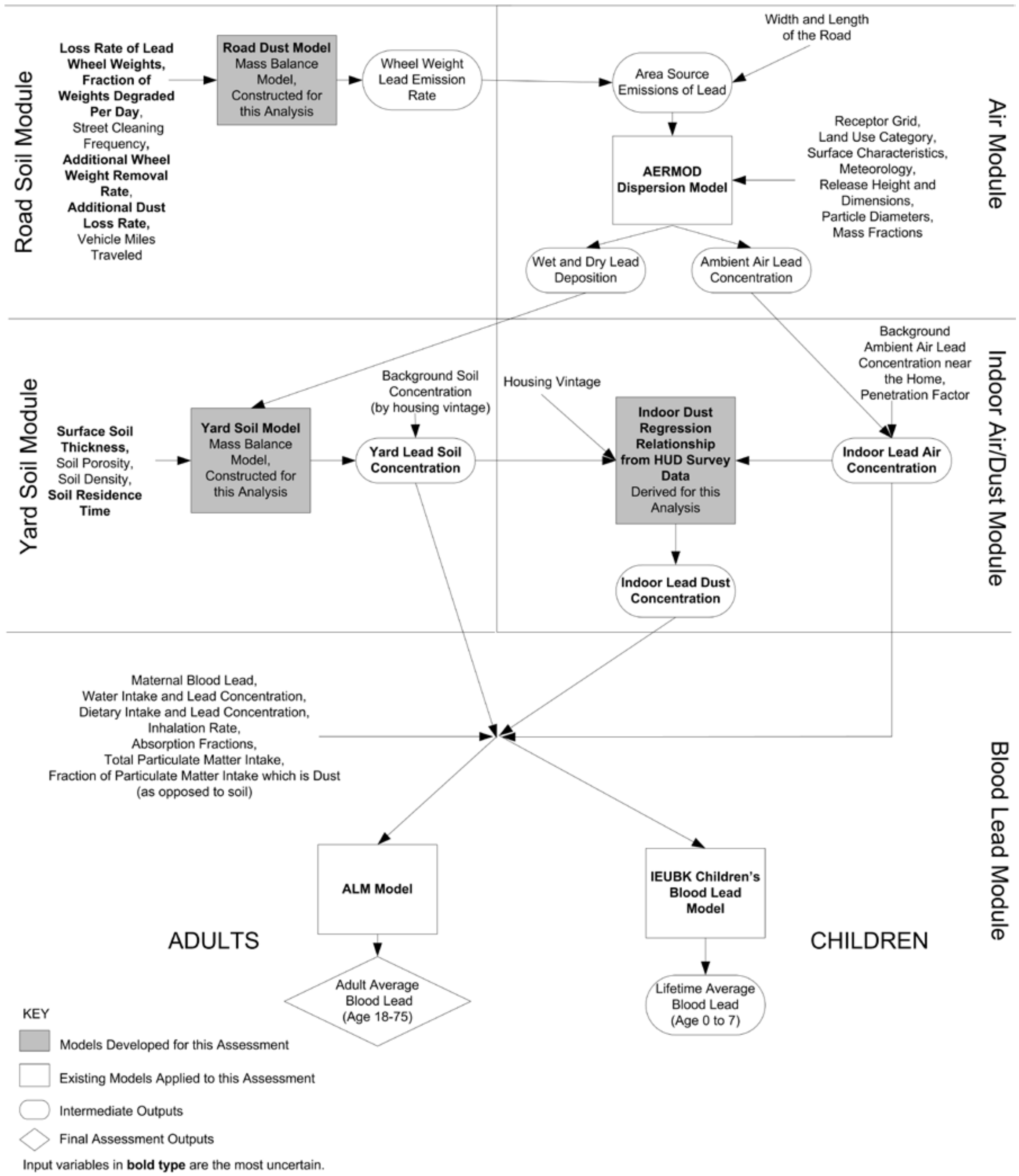

Figure 3. Flowchart Showing the Assessment Approach for the Near-Roadway Residence Exposure Scenario

4.1 Roadway Soil Module

The roadway soil module estimates the total emission rate of lead dust from the roadway for each modeling scenario, as depicted in Figure 4. Section 4.1.1 discusses the assessment method selected for this module, while Section 4.1.2 discusses how each input parameter value was selected.

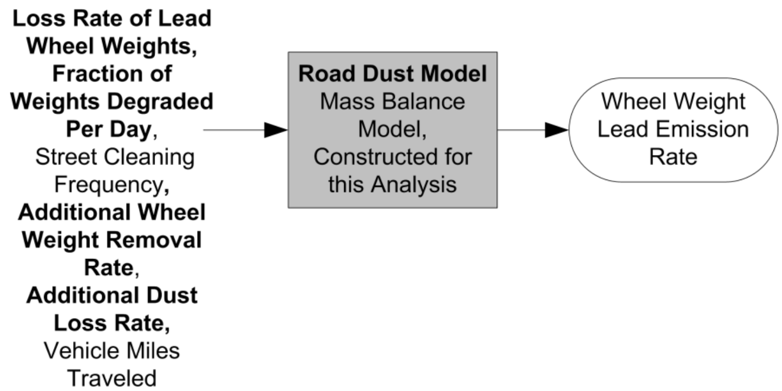

Figure 4. Flowchart Showing the Approach for the Roadway Soil Module

4.1.1 Assessment Method Selected

Wheel weights are lost from cars onto the road, and this loss rate is dependent on the traffic flow rate, the proportion of traffic vehicles that have lead wheel weights, the speed, and the degree to which the road requires braking and turning events. Then, lost wheel weights are degraded over time due to weathering and further traffic abrasion. Some of the lead that is abraded will be emitted to the air as part of roadway dust due to roadway turbulence and other dust emission mechanisms.

A literature search located a peer-reviewed article looking at lead wheel weight degradation, the Root (2000) study. This study estimates (i) the baseline or steady-state inventory of lead wheel weights on an urban street (ii) the average loss rate of lead wheel weights from passing automobiles and (iii) the average rate of lead wheel weight fractional degradation per day as a result of abrasion and pulverization by moving vehicles on the street. The study is based on measurements conducted on a 2.4km (1.5 mile) six-lane divided street segment in Albuquerque, New Mexico with an average daily traffic flow of 41,500 vehicles and a reported speed limit of 65 km/hour (40 miles/hour). Topographically, the street, which is identified in the study by the letters “JTML”, is characterized by a slightly elevated crown in the middle of the street that slopes off to curbs at either side to facilitate storm water drainage. To estimate the steady-state inventory on the street, the author surveyed JTML and seven other streets in the same city by walking along the sidewalk adjacent to the outer lane and collecting any lead found along the outer curb, in the street, and on the sidewalk. In some segments, the sidewalk was set back from the curb with the intervening space occupied by gravel and shrubs. The author reports that these obstacles made searching for wheel weights more difficult. Curbside parking did not occur on the street in the surveyed area. The author conducted only one survey along the median end of the divided street because of the potential danger from passing vehicles. The cleaning history of the streets is reported as not known. Based on the inter-street consistency of the amount of lead found on the eight streets studied, the study concludes that the streets were in a steady-state condition. The author resurveyed two of the eight street segments to ensure that steady-states were consistent over time. Based on this method, a mean steady state inventory of 1.09 kg/km is reported for JTML. The study notes that the quantity of lead deposited may be underestimated because of the difficulty involved in ensuring complete recovery of all lead pieces. The study also reports that wheel weights along the median end of the street were 25% of the steady state.

The study estimates the average loss rate of lead wheel weights from passing automobiles and also the average rate of lead wheel weight fractional degradation per day by means of biweekly surveys conducted on the same JTML street segment for 42 weeks. The biweekly surveys were conducted using exactly the same process as the steady state inventory analysis described above. The amount of lead collected in each biweekly study represents the accumulated lead after 14 days of successive deposition less the amount pulverized in that period. By assuming a constant daily average loss rate from automobiles and a constant average daily fractional degradation rate, the study derives algebraic relationships that enable the estimation of the average loss rate and the fractional degradation rate per day from knowledge of the steady state inventory and the average biweekly inventory. Using this method, the study estimates the average deposition rate of lead wheel weights along the outer curb of both sides of JTML street as 11.8 kg/km/year and the average fractional degradation rate per day as 0.0272 (or 2.72%).

Given the limited sample of streets surveyed, precautions may also be advisable when extrapolating the findings to other roadway environments. Roadways with higher speed limits and with more roadway irregularities (such as pot holes) than the JTML street will be more likely to cause ejection of wheel weights onto the roadway. Also, the study assumes all missing wheel weights have been pulverized; it does not account for loss processes such as removal during street cleaning, collection by hobbyists or dispersal outside the area of the survey. A consequence of this assumption is that the estimated fractional degradation rate is in effect a fractional loss rate owing to all loss processes, which could represent a potential upper bound for the true fractional degradation rate. The use of the estimated fractional degradation rate could potentially result in estimates of risk from wheel weight-derived roadway lead dust that are biased high. The study estimates could also be biased (inaccurate) as a result of measurement error. The author concedes that the collection process may have overlooked some lead wheel weights fragments on the road. While there is the possibility that proportional measurement errors in the biweekly surveys and the steady state survey could cancel out, resulting in an unbiased estimate of the fractional degradation rate, it is also conceivable that the measurement errors in each type of survey were not proportional and could potentially result in a biased estimate of the fractional degradation rate. In addition, measurement error in the steady state inventory estimate could result in a biased estimate of the average loss rate. The logistic constraint that excluded collection from the median curbs of the divided street or from the central parts of the roadway is an additional source of uncertainty. Extrapolating the study's outer curb loss rate to derive a "whole street" loss rate would consequently create a further source of potential bias. The study is based on a single street segment in a single city. Finally, the study is deterministic in nature and provides only point estimates without any confidence intervals to bound potential variability or uncertainty.

Another study (Bodanyi, 2003) has not been published in a peer-reviewed publication. It was conducted by the author as part of a student thesis. One of the principal aims of the Bodanyi study appears to be a comparison of the author's estimate of lead wheel weight deposition onto urban roadways with the earlier published study by Root. The Bodanyi study was conducted on two thoroughfares in Ann Arbor, MI and estimates the number of lead wheel weights lost per vehicle mile traveled (VMT) on urban roadways. The study employed the same visual survey and recovery methods as Root (2000), but was limited to four weekly surveys. Based on the recovery rate from these surveys, the Bodanyi study estimates that 4.69E-5 wheels weights are lost per VMT. The study uses this loss rate to estimate the total deposition of lead onto U.S. highways at 2.7 million kg in the year 2001. According to Bodanyi, the Root study may be inferred to estimate a loss rate of 4.58E-5 wheel weights per VMT by assuming that the average wheel weight recovered from the roadway weighs 21 g (as reported by Root). This estimate indicates strong agreement between the Root and Bodanyi studies. However, in translating the lead wheel weights per VMT into the mass of lead deposited per VMT, the Bodanyi study appears not to account for roadway degradation. Bodanyi simply multiplies the estimated number of wheel weights deposited per VMT by the average weight of a *recovered* wheel weight to estimate the mass of lead wheel weights deposited per VMT. This is likely an underestimate because a recovered wheel weight has already been abraded to a certain extent. The Root study employs sounder principles in accounting for the effect of wear while estimating lead deposition. These data were not used in this exposure assessment but mentioned to show the corroboration that wheel weights are found on the street in degraded states.

A poster presented at the Geological Society of America, on 31 October – 4 November 2010, provided information on an on-going roadside wheel weight collection study being conducted by students in University of West Georgia under Professor Curtis Hollabaugh. Preliminary results of this program indicate that many wheel weights are found along urban and rural roads in Georgia. Many of these lead wheel weights are small, worn and weathered; several were small and without clips. Although quantitative data on degradation rates are not yet available from this program, it shows that wheel weights are falling onto the several roads in urban, suburban, and rural Georgia and are being degraded.

The loss rates and degradation rates calculated in the Root (2000) study are used in this analysis. It should be noted that wheel weights are 95% lead and 5% antimony. No attempt has been made to correct the degradation rate to include only the lead portion, since the error introduced into the analysis by assuming the wheel weight is 100% lead is small compared with the overall uncertainty in the loss and degradation rates. With the exception of street cleaning rates, information about the other wheel weight removal rates could not be found in the literature, but the module addresses them as discussed below.

The most general method of modeling lead emitted to the air as part of roadway dust would be by tracking the mass of intact wheel weights, the mass of lead dust on the roadway, and the mass of roadway dust emitted each day by accounting for time-varying source and loss rates. For this analysis, it was assumed that the system is in a nearly steady-state. The near-steady state assumption implies that the mechanisms dictating the accumulation of wheel weights on each segment of roadway (including all sources and removal mechanisms) are in balance so that the total inventory of wheel weights along the road segment is not changing in time. In this analysis, a road segment is set to one city block. In addition, the amount of lead dust generated each day from degradation would equal the sum of the removal due to emission and the removal due to other loss rates like runoff. The Root (2000) study observes that steady state conditions are rapidly achieved on a roadway; empirical calculations made for this analysis also support this conclusion, with steady state conditions typically being reached within one year.

Lead dust emissions have therefore been computed at "average" steady state conditions using a mass balance model designed for this analysis. The mathematical relationship between the fall off rate of intact wheel weights and the average lead dust emission rate at steady state conditions was derived as follows:

Let:

F = the loss (fall-off) rate of intact lead wheel weights from cars onto the roadway (in kg per day)

X = the mass of intact lead wheel weights on the roadway (in kg)

Y = the mass of lead dust (originating from the degradation of lead wheel weights) on the roadway (in kg)

d = degradation rate (the fraction of lead wheel weights that are converted to lead dust per day)

u= street cleaning rate (the fraction of lead wheel weights that are removed from the road per day)

h= the loss of partially intact wheel weights to loss mechanisms other than degradation and street cleaning

l = the loss of degraded mass due to loss mechanisms other than emission

e = emission rate (the fraction of roadway lead dust that is suspended into the air by vehicles per day)

Mass balance considerations dictate that:

(1) the change in mass of lead wheel weights on the roadway on a given day will equal the mass of lead wheel weights falling off from cars onto the roadway that day less the mass of wheel weights degraded into dust on the roadway that

day less the mass of wheel weights removed by road cleaning and other loss mechanisms that day (see Figure 5); and

(2) the change in mass of lead dust on the roadway on a given day will equal the mass of lead dust added to the roadway that day by degradation less the mass of lead dust suspended into the air by passing automobiles on that day and the mass of lead lost due to other mechanisms.

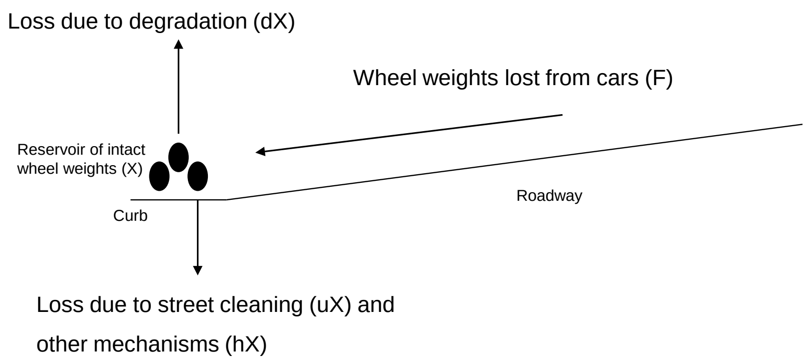

Figure 5. Diagram of the processes governing the stock of wheel weights in the curb

Using the symbols defined above, these mass balance equations may be expressed mathematically in terms of the following differential equations:

$$\frac{dX}{dt} = F - dX - uX - hX \qquad (1)$$ and

$$\frac{dY}{dt} = dX - eY - lY \qquad (2)$$

At steady state, the mass of intact wheel weights (X) and the mass of lead dust (Y) are constant, implying that $\frac{dX}{dt} = \mathbf{0}$ and $\frac{dY}{dt} = \mathbf{0}$ ..

Setting $\frac{dX}{dt} = \mathbf{0}$ and $\frac{dY}{dt} = \mathbf{0}$ in equations (1) and (2) above, results, after some algebraic manipulation, in the following steady state relationships:

$$X_{ss} = \frac{F}{d + u + h} \qquad (3)$$

$$\frac{eY_{ss}}{F} = \frac{d}{d + u + h}\,\frac{e}{e + l} \qquad (4)$$

Equation (4) illustrates how the steady state emission of lead dust to the air from the roadway (eY_{ss}) is a fraction of the loss rate of intact lead wheel weights onto the roadway (F). If street cleaning and the additional loss terms do not exist (u, h, and l), then at steady state the emission of lead dust equals wheel weight deposition on the roadway.

A complication that prevents a purely analytic estimation of the steady state emission rate of lead dust is that the street cleaning rate u in the equation above is not a constant but varies with time (to reflect the reality that street cleaning occurs not continuously but at a periodic frequency). For a street with a monthly cleaning frequency, it was assumed that u would equal zero for days 1-29 and then equal 1 on the thirtieth day, after which it would assume the value zero for the next 29 days, and so on. This assumes that street cleaning removes the entire stock of wheel weights on the curb on the days that it occurs. Consequently, the average steady state emission rate was estimated empirically using a dynamic spreadsheet model that directly simulates equations (1) and (2) above.

The occurrence of cyclical street cleaning prevents the realization of a true unvarying steady state; instead a "cyclical steady state" is achieved in which the emission rate and other variables repeat the same values on a cyclical basis related to the cleaning frequency. Figure 6 shows the wheel weight loss rate as well as the cyclical dust emission rate for the urban scenario. For the purposes of computing average exposure and risk, the average dust emission rate across the cycles was used. The ratio between the wheel weight loss rates and the average dust emission rate for each scenario was computed and used along with the estimated loss rates to estimate the total lead mass emission rate.

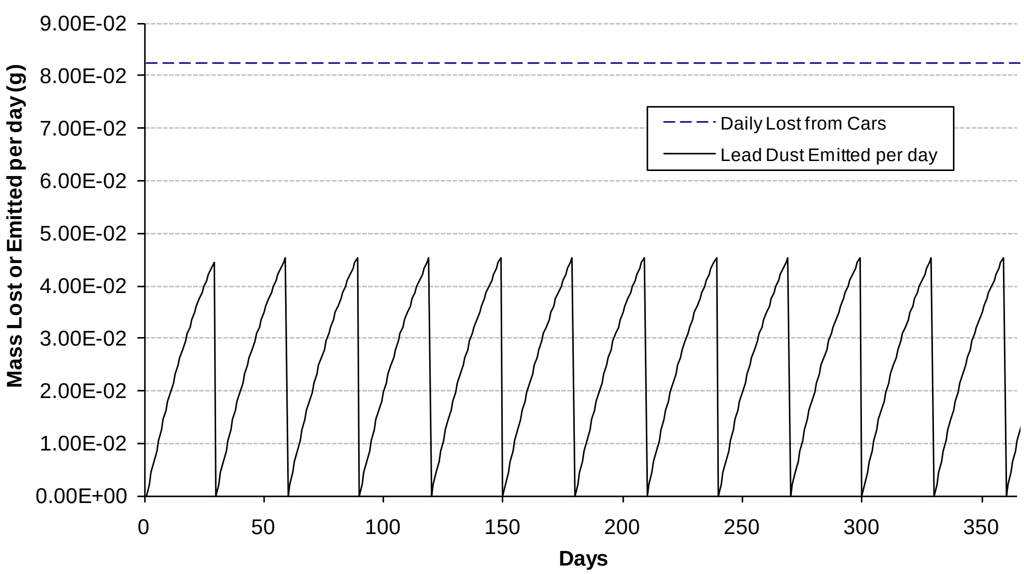

Figure 6. Mass of Wheel Weights Emitted Per Day in a 1 m Urban Segment of Road and the Cyclical Lead Dust Emitted from the Roadway Each Day

4.1.2 Parameter Selection

Loss Rate of Lead Wheel Weights

The loss rate of lead wheel weights is derived from information presented in Root (2000). The study estimates wheel weight lead deposition along the 2.4 km six-lane divided "JTML" road in Albuquerque, New Mexico at 11.8 kg/km/year. The study notes that this estimate represents the loss along the outer curb of both sides of the street. The study also observes that the median side deposition amounts to 25% of the curb side loss at steady state. To include loss along the median edge of both sides of the divided street, the curb side loss rate estimate was multiplied by a factor of 1.25 in order to estimate the loss rate for the entire street. Accordingly, it was assumed that the lead wheel weight loss rate was 1.25 x 11.8 = 14.75 kg/km/year, which is equivalent to 23.6 kg/mile/year along that street segment. To normalize the lead wheel weight loss rate by the vehicle miles traveled, an average daily traffic flow of 41,500 vehicles/day was used, which is the traffic flow rate for the surveyed JTML street segment as cited in the Root study. The estimated normalized wheel weight lead loss rate is therefore equal to 23.6/(41500 x 365) = 1.56 E-6 kg/VMT. This loss rate was multiplied by the vehicle counts discussed below to estimate the total mass per mile traveled.

Causes of other variations in the loss rate, such as the speed of traffic on the road and the mix of vehicles on the road, could not be accounted for since there was not enough information in the literature to inform a methodology. In the Root study, the speed limit was 65 km/hour or about 40 mph. This will be similar to high-traffic residential roads in the proxy cities but will overestimate the speed of traffic on the low-traffic residential roads (see Section 4.2.2, "AERMOD Grid"). Thus, the loss rate on the low-traffic roads may be overestimated, since more wheel weights will be lost to cars when they turn or hit pot holes and other bumps at higher speed. However, lower traffic roads may also have more bumps and road imperfections, as higher volume roads will be given priority for repairs. Thus, the effect of the speed cannot be determined from the existing information in the literature and is not accounted for in this analysis approach. The wheel weight loss rate remains an uncertain variable and is examined in the uncertainty analysis in Section 4.9.

Fraction of Weights Degraded Per Day

The fraction of lead wheel weights degraded per day is also obtained from the Root (2000) study, where it is estimated at 0.0272 or 2.72%. Although the Root study has numerous limitations, including the fact that the only loss mechanism considered was loss to degradation, no superior study on the subject could be found despite an extensive literature search. In the absence of any additional information about the particular street used in the study (including the cleaning history) and information about other loss processes in general, a daily degradation rate of 2.72% was used and the other loss processes were accounted for as described below. This may lead to an overestimate of the total loss of intact wheel weights; however, in the absence of data, this method was selected as the most systematic one available. The wheel weight degradation rate remains an uncertain variable and is examined in the uncertainty analysis in Section 4.9.

Street Cleaning Frequency

To determine the typical frequency of street cleaning, statistics of street cleaning from various cities were pulled from a compiled report (Schilling, 2005). The statistics show the frequency of street cleaning for a main artery, a central business district, and a residential area. Because the modeling domain includes the intersection of two busy streets in the urban, suburban, and rural scenarios and the highest concentration occurs at the crossroads (see Section 4.7), the central business district statistics were selected as the most appropriate descriptor of cleaning frequency. These frequencies are higher than in the purely residential area but reflect probable cleaning frequencies for high volume roads near residential areas.

For each city, the population, population density, and city type were determined from census information. The population corresponds to census information from 2006 while the population density (persons per square mile) corresponds to census information from 2000 (http://quickfacts.census.gov/qfd/). Using the population and population density, each city was mapped to a city type using the following census definitions:

- Urban Area (UA): 500 people per square mile with at least 50,000 people.
- Urban Cluster (UC): 500 people per square mile with a population of at least 2,500 people, but fewer than 50,000 people.
- Rural: anything outside of the definition of UC or UA

If a city did not have available population density information, the population alone was used to map the city to a classification. Then, the UA designation was used to capture the urban areas of modeling scenarios A and B, the UC designation was used to capture the suburban areas in modeling scenario E, and the rural designation was used to capture the rural areas in modeling scenarios C and D. In the dataset used, no cities had the rural designation, and the classifications of each city are shown in Table 2.

Then the frequencies of cleaning were averaged across each classification category to determine the average number of days between cleaning. These averages were rounded to regular frequencies. This resulted in a frequency of once every month in urban areas and six times per year in suburban areas. In the absence of any rural information, a cleaning frequency of two times a year, which is the lowest frequency reported in the survey, was selected for these locations. It was assumed that street cleaning has a 100% efficiency in removing wheel weights such that the entire reservoir of wheel weights along the curb is eliminated after each street cleaning event.

Table 2. Street Cleaning Statistics and City Classifications

City	State	Arterial	Central Business District	Residential	Population	Population Density	Classification
Oakland	CA	Daily		Biweekly	397,067	7,126	Urban Area
San Diego	CA		Weekly	Monthly	1,256,951	3,772	Urban Area
San Leandro	CA			Monthly	78,030	6,051	Urban Area
Long Beach	CA	Weekly	Weekly	Weekly	472,494	9,150	Urban Area
Mountain View	CA			Biweekly	70,090	5,863	Urban Area
San Jose	CA	Biweekly	Biweekly	Monthly	929,936	5,118	Urban Area
La Mesa	CA	2x/week	2x/week	Monthly	53,043	5,912	Urban Area
Sunnyvale	CA			Monthly	130,519	6,006	Urban Area
Union City	CA	Biweekly	Biweekly	Biweekly	69,477	3,474	Urban Area
Danville	CA	Monthly	Monthly	Monthly	41,540	2,306	Urban Cluster
Dublin	CA		Weekly	Biweekly	41,840	2,381	Urban Cluster
Elk Grove	CA	Monthly		3x/year	129,184	No data	Urban Area
Santee	CA	Weekly	Weekly	Biweekly	52,530	3,299	Urban Area
Greeley	CO	Biweekly	Weekly	5x/year	89,046	2,573	Urban Area
Fort Collins	CO		2x/week	2x/year	129,467	2,550	Urban Area
Denver	CO		Biweekly	8x/year	566,974	3,617	Urban Area
Thornton	CO	Biweekly		1x/year	109,155	3,067	Urban Area
Arvada	CO	6x-7x/year	6x – 7x/year	6x-7x/year	104,830	3,128	Urban Area
Tampa	FL	Weekly	Weekly	6x/year	332,888	2,708	Urban Area
Gainesville	FL	Monthly	2x/week	9x/year	108,655	1,981	Urban Area
Urbandale	IA	3x/year	3x/year	3x/year	37,173	1,405	Urban Cluster
Iowa City	IA	Monthly	Weekly	Monthly	62,649	2,575	Urban Area
Sioux City	IA	5x/year	5x/year	5x/year	83,262	1,551	Urban Area
Overland Park	KS	7x/year	Monthly	3x/year	166,722	2,627	Urban Area
Hanover Park	IL	8x/year	8x/year	8x/year	37,161	5,637	Urban Cluster
Evanston	IL	Biweekly		4x/year	75,543	9,579	Urban Area
Elgin	IL	Biweekly	2x/week	6x/year	101,903	3,780	Urban Area
Burr Ridge	IL	9x/year	9x/year	9x/year	10,408		Urban Cluster
Champaign	IL		Daily	8x/year	73,685	3,974	Urban Area
Fort Wayne	IN	Biweekly	Weekly	4x/year	248,637	2,606	Urban Area
Cambridge	MA	Biweekly		9x/year	101,365	15,763	Urban Area
Salem	MA			9x/year	41,343	4,989	Urban Cluster
Saco	ME	Biweekly		9x/year	16,822		Urban Cluster
Kansas City	MO	4x/year	Weekly	4x/year	447,306	1,408	Urban Area
St. Joseph	MO	2x/year	2x/year	2x/year	72,651	1,688	Urban Area
Great Falls	MT	Biweekly	Daily	4x/year	56,215	2,909	Urban Area
Lincoln	NE			3x/year	241,167	3,022	Urban Area
Manchester	NH	Monthly	2x/week	3x/year	109,497	3,242	Urban Area
Albuquerque	NM	Biweekly	2x/week	Biweekly	504,949	2,483	Urban Area
Rochester	NY	2x/week	Daily	Biweekly	208,123	6,134	Urban Area
Albany	NY	Weekly	Weekly	Weekly	93,963	4,474	Urban Area
Toledo	OH	9x/year	2x/week	9x/year	298,446	3,890	Urban Area
Fairfield	OH	Biweekly	Weekly	5x/year	42,248	2,006	Urban Cluster

Table 2. Street Cleaning Statistics and City Classifications

City	State	Arterial	Central Business District	Residential	Population	Population Density	Classification
Macedonia	OH	2x/year	2x/year	2x/year	9,224		Urban Cluster
Marysville	OH	Weekly	Weekly	Monthly	18,212		Urban Cluster
Tulsa	OK	8x/year		4x/year	382,872	2,152	Urban Area
Albany	OR	Biweekly	Weekly	Monthly	46,213	2,573	Urban Cluster
Eugene	OR	Weekly	2x/week	Monthly	146,356	3,403	Urban Area
Pittsburg	PA	Weekly	2x/week	2-4x/year	312,819	6,020	Urban Area
Town of Lower Marion	PA	3x/year		3x/year	59,850		Urban Area
Knoxville	TN		Weekly	Monthly	182,337	1,876.60	Urban Area
San Antonio	TX	4x/year		2x/year	1,296,682	2,809	Urban Area
Dallas	TX	Monthly	Daily	None	1,232,940	3,470	Urban Area
El Paso	TX	Biweekly	Daily	4x/year	609,415	2,263	Urban Area
Austin	TX		Daily	6x/year	709,893	2,610	Urban Area
Ogden	UT	3x/year	3x/year	3x/year	78,086	2,898.90	Urban Area
Hampton	VA	Monthly		Monthly	145,017	2,828	Urban Area
Janesville	WI		5x/year	4x/year	62,998	2,160	Urban Area
Eau Claire	WI	3x/year	3x/year	3x/year	63,297	2,037.80	Urban Area
Milwaukee	WI		Weekly	Monthly	573,358	6,215	Urban Area

Additional Intact Wheel Weight Removal Rate

Aside from street cleaning, partially intact wheel weights are removed from the roadway due to other mechanisms. Hobbyists may gather wheel weights from along the roadway. In addition, weights may be thrown into the median or into grassy areas and thus protected from further roadway abrasion. Ignoring the impact of these loss mechanisms will tend to an overestimate of the risks from lead wheel weights. However, there are no data available to inform the decision of the fraction removed and so the fraction of lead wheel weights lost due to these mechanisms per day was set at zero (0)   This variable remains highly uncertain, and it is examined in the uncertainty analysis in Section 4.9.

Additional Roadway Dust Loss Rate

Once wheel weights have been degraded, the lead remains on the roadway and curb as lead dust. Some of this dust will be emitted to the air due to wind and the turbulence generated by passing vehicles. However, this dust will also be removed from the roadway due to water runoff or other loss processes. During rain events, this removal may be significant. However, because the literature data provide no way to determine this fractional removal, this loss rate is set to 0 to represent a high-end estimate. This variable remains highly uncertain, and it is examined in the uncertainty analysis in Section 4.9. In addition, speciation can make lead more or less toxic, but this process is highly variable. Due to the complexity of factors that determine speciation, this process was not included.

Vehicle Miles Traveled

The total traffic counts for each scenario are shown in Table 3. In each scenario, the model grid consists of a series of intersecting roads. Roads are either designated as "high volume" or "low volume". The urban traffic counts were determined based on examination of traffic counts in the Northeast proxy city provided by the state department of transportation. For the high volume streets, a busy, four lane road near residences was selected (33,800 vehicles per day). The traffic counts on a strictly residential road (low volume) in the same proxy city were also determined, and the ratio between the low traffic street and the high traffic street was approximately 0.25. In addition, the higher volume streets occurred approximately every kilometer with lower volume streets between them. Thus, the urban model domain consists of a series of intersecting high volume streets with 33,800 vehicles per day every kilometer in both the north/south and east/west directions with lower volume streets with 8,450 vehicles per day spaced in the intervening blocks.

For the rural scenarios, traffic counts in a western proxy rural community were used to determine the traffic counts. The only available data were for a relatively high volume street through the town (755 vehicles per day). No data were available for the lower volume residential roads in the rural community. Thus, the same ratio between low volume and high volume streets used in the urban and suburban scenarios (0.25) was used to estimate a volume of 189 vehicles per day on low volume streets.

Traffic counts in a northeastern proxy suburban community were used to determine the traffic counts for the suburban scenario. The traffic count for the highest traffic volume street near residences (3,100 vehicles per day) was selected. In addition, the ratio between a lower volume residential street and this high volume street were determined to be approximately 0.25. Thus, the same ratio between low volume and high volume streets used in the urban and rural scenarios (0.25) was used to estimate a volume of 775 vehicles per day on low volume streets.

Table 3. Estimate of Average Daily Traffic Counts by Road Type for Each Scenario.

Scenario	High Traffic Volume Average Daily Traffic (vehicles/day)	Low Traffic Volume Average Daily Traffic (vehicles/day)
Urban	33,800	8,450
Rural	755	189
Suburban	3,100	775

Once the cleaning frequency and loss rates were determined for each scenario, the average steady-state mass balance model was applied to each scenario. Table 4 presents empirically computed ratios of average steady state roadway lead dust emission rates to lead wheel weight fall off rates. To estimate the final lead emission rates, the wheel weight fall off rate (1.56 E-6 kg/VMT, see above) was multiplied by this ratio and by the vehicle counts on the individual roads (high volume and low volume) in the domain for each scenario to get the total mass emitted per day.

Table 4. The Ratio of Lead Dust Emission to Wheel Weight Fall Off Rates in Urban, Suburban and Rural Areas

Scenario	d (Degradation Fraction per day)	Cleaning Frequency (in days)	$\frac{eY_{ss}}{F}$ (Ratio of Average Steady State Emission Rate of Lead Dust to Loss Rate of Wheel Weights)	Emission Rates on High Volume Streets (g m^{-2} s^{-1})	Emission Rates on Low Volume Streets (g m^{-2} s^{-1})
Urban	0.0272	30	0.31	1.47E-8	3.66E-9
Suburban	0.0272	60	0.53	2.30E-9	5.75E-10
Rural	0.0272	183	0.81	8.56E-10	2.14E-10

4.2 Air Module

In order to characterize the air concentrations and depositions resulting from the roadway lead wheel weight emissions, a dispersion model was needed. The method selected and the necessary input parameters are depicted in Figure 7. Section 4.2.1 discusses the assessment method selected for this module, while Section 4.2.2 discusses how each input parameter value was selected.

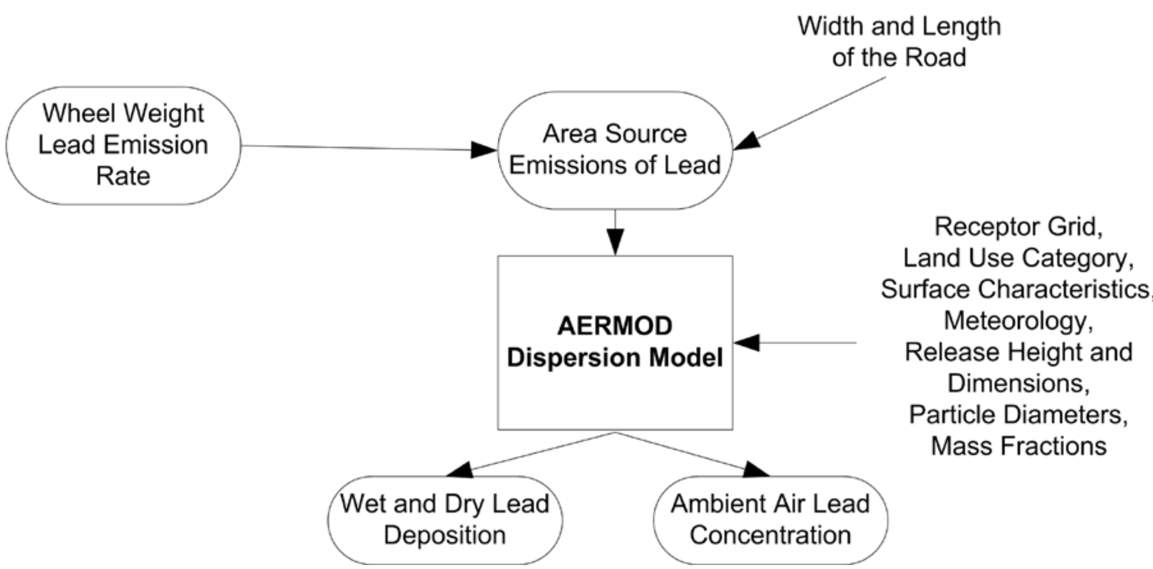

Figure 7. Flowchart Showing the Approach for the Air Module

4.2.1 Assessment Method Selected

In order to model the dispersion of wheel weight lead away from the roadway to neighboring homes, the AERMOD dispersion model was selected (U.S. EPA, 2009a).

According to the “Revision to the Guideline on Air Quality Models” (U.S. EPA, 2005b), AERMOD represents the most robust air quality model when evaluated against monitoring data. A multimedia model, the TRIM.FaTE model, was also considered since it allows explicit communication between air and soil compartments and would not require a separate yard soil module (see Section 4.3). However, TRIM.FaTE does not have as sophisticated a dispersion scheme, so AERMOD was selected to best capture the yard air concentrations.

Once the AERMOD model was selected, model options had to be selected in order to model the roadway dust emission and dispersion. To implement AERMOD, the modeled city was assumed to consist of a series of streets that intersect at regular intervals. Based on proxy cities for the urban, rural, and suburban scenarios, the block length, street width, and number of houses per block were used to create the emission grid (the roadways) and the receptor grids (individual yards). To account for different traffic patterns within a city, the grid contains both main arteries and residential streets, where each occurs at specified regular intervals (see Section 4.1.2). Then, road sources were modeled in AERMOD as area sources. According to “Revision to the Guideline on Air Quality Models”, re-entrained dust from roadway sources can be modeled as area, volume, or line sources (U.S. EPA 2005b, page 68235). Area sources were selected to be consistent with the OTAQ lead in aircraft exposure analysis, which modeled three roadways adjacent to the airport using this methodology (U.S. EPA 2010b, page 49). No obstructions due to the presence of other buildings were included in the modeling, since this introduces a level of detail to the modeling that the input data quality did not support. Obstructions can both enhance air concentrations and diminish air concentrations depending on the location of the model receptor with respect to the emission site and the obstruction. Instead, the surface characteristics needed for input into the model were determined to be consistent with the typical building characteristics in each scenario, as explained in Section 4.2.2 “Land Use Category and Surface Characteristics”. The roadway dimensions, traffic patterns, and lead emissions are combined to estimate the area source of lead from the roadway (see Table 4). This source represents the source of lead-containing dust which is lifted from the road surface due to turbulence due to passing traffic and then subsequently dispersed. Meteorological conditions, land use information, and particulate attributes are also input into AERMOD for the dispersion calculation. The model was run for a single year, and this year was considered to be representative of a typical year during the life of the child or adult. The outputs of this module are the estimated annual-average ambient air concentration, dry lead deposition, and wet lead deposition at each receptor (i.e., individual yard).

4.2.2 Parameter Selection

AERMOD Grid

For each scenario, the traffic volume and street grid were determined using general attributes of urban, suburban, and rural cities, as described below. In each case, the proxy city was used to represent a typical city. The grid included high volume roads and low volume roads, and the grid was constructed as discussed below.

For the urban scenarios, the proxy city is Dorchester, MA, which is characterized by multifamily homes and small yards. Measurement tools in GoogleEarth® were used to examine three different blocks in the center of Dorchester to determine that a typical block is rectangular with the dimensions 150 x 60 m and the streets are 8 m wide. Visual inspection in GoogleEarth® revealed that there are typically 8 yards x 2 yards per rectangular block. High traffic volume streets occur approximately every kilometer, with low volume streets occurring in between these using the block length as the spacing.

Then, a series of runs were performed using different total modeling domain sizes to determine how far the grid of source streets should extend to capture the full contribution of wheel weights at the home of highest air concentration. This home occurs at the intersection of two busy streets near the center of the domain. Initially, a grid size of 3 km was used. Then, because the wind direction is predominantly from the west-northwest (see below), an additional high volume street was added in the western direction, bringing the total extent in the east-west direction to 4km. In this case, the maximally exposed home increased by 4%. However, the addition of more receptors and street area sources greatly increased the runtime. Thus, the concentrations from the 3km run were used in the analysis. The 4km run suggests that these estimates could be under-predicting the lead concentration by up to 4% or more, and given the uncertainty in the emission rates, this amount of difference was deemed acceptable for this modeling effort.

For the rural scenario, the western rural community of Boulder, MT was used as the proxy city. Measurement tools in GoogleEarth® were used to examine the center of the city and one representative block to determine that a typical block is square with the dimensions 115 x 115 m and the streets are 8 m wide in the downtown area. Visual inspection in GoogleEarth® revealed that there are typically 3 yards x 2 yards per square block. The extent of the rural community was approximately 1 km with only a single high volume intersection. Thus, the model domain extended 1 km in the north/south and east/west directions with a single high volume intersection in the middle of the domain, with lower volume streets spaced in between at intervals equal to the block length. No sensitivity test was done to increase the grid size, since it was deemed unlikely a rural town would extend further than 1 km.

Finally, for the suburban scenario, the northeast suburban city of Turners Falls, MA was used as the proxy city. Measurement tools in GoogleEarth® were used to examine the center of the city and one representative block to determine that a typical block is rectangular with the dimensions 200 x 105 m and the streets are 8 m wide in the downtown area. Visual inspection in GoogleEarth® revealed that there are typically 5 yards x 2 yards per square block. Inspection of the pattern of roads indicated that higher volume streets occurred every 1 km in the suburban community. Thus, the domain consisted of a 2 km square with an intersection of higher volume roads in the center and higher volume roads along the perimeter, with lower volume roads along the intervening blocks. Owing to the lower emission rates from wheel weight dust release in this scenario, this domain size was sufficient to capture the air concentration and deposition estimates.

To estimate the area source emission rates, the lead emission rates (see Table 4) from the road soil module were multiplied by a factor of 1E8 (urban scenarios) or 1E9 (suburban and rural scenarios) to allow increased modeling precision. This factor was then divided out when calculating the modeled air concentrations and depositions at the maximally exposed home.

Land Use Category and Surface Characteristics

AERMOD (specifically, the meteorological preprocessor, AERMET) requires the land use distributions of the study sites in order to estimate monthly values of three important surface characteristics (surface roughness length, albedo, and Bowen ratio). AERMOD's land-use preprocessor, AERSURFACE, was developed to read in National Land Cover Database (NLCD) land use data (version 1992), calculate the distribution of land use types surrounding the study site, and use look-up tables where the values of the three surface characteristics depend on land use, season, snowfall, and rainfall amount. These surface characteristic look-up tables are available in Appendix A of the AERSURFACE User's Guide (U.S. EPA, 2008a). However, this study models a simplified grid of city blocks that each have the same land use characteristics within the same scenario (within the urban scenario, for example), rather than more realistic heterogeneous land use. As such, certain land use aspects of AERSURFACE (e.g., setting a land use radius for the surface roughness length, setting unique land use sectors) are not needed. Instead, the distribution of land use types surrounding the study sites was manually estimated, and, after also determining the climate characteristics, the look-up tables from U.S. EPA (2008a) were used to estimate the values of the three surface characteristics.

The land area covered by residential buildings was estimated by first estimating the ground footprint of the typical residential building at each study site in this study (urban, suburban, and rural). Residential buildings include apartment buildings and attached and detached single family homes. The 2005 Residential Energy Consumption Survey results from the U.S. Energy Information Administration (U.S. E.I.A., 2005) were used to estimate these footprints. Table 5 shows the estimated national number of the various types of residence buildings, the estimated percentage of each of these buildings at each of the study sites, and the estimated national average footprint of these buildings. The final column in Table 5 shows the assumptions that were made to estimate these numbers for this study. Note that towns are not used in this study but are shown in the table for completeness.

Table 6 shows the estimated average residence building footprint at each of the study sites. All of the footprints are between 190 and 205 m^2 (2,000 and 2,200 ft^2). Cities have the largest average footprint (203 m^2) due to a higher percentage of apartment buildings relative to single family homes, while rural areas have the smallest average footprint (193 m^2) due to a very small percentage of apartment buildings.

Assuming that urban residential buildings tend to be taller than rural and suburban residential buildings, residential buildings for the urban study site were linked to the land use type "High Intensity Residential" (USGS, 2010). Residential buildings for the rural and suburban study sites were linked to the land use type "Low Intensity Residential" (USGS, 2010). These land use designations are shown in Table 7. Table 7 also shows that cumulative footprint of residence buildings per city block, which was calculated by multiplying the average residence building footprint (Table 6) by the number of yards per city block. The cumulative footprint of residence buildings per city block ranges from about 1,160 m^2 at the rural study site to about 3,251 m^2 at the urban study site.

Table 5. Estimated U.S. Residence Building Characteristics

	National Total Count	% of National Total in…				Avg Footprint (m^2)	Assumptions
		Cities	Suburbs	Towns	Rural Areas		
Detached Single Family Homes, 1 Floor	53,300,000	33%	22%	17%	27%	209	Detached single family homes include mobile homes, split-level, and 'other'
Detached Single family Homes, 2 Floors	24,000,000					161	
Detached Single family Homes, 3+ Floors	1,700,000					130	All have only 3 floors
Attached Single Family Homes, 1 Floor	2,600,000	64%	20%	16%	N/A	209	
Attached Single family Homes, 2 Floors	4,000,000					161	
Attached Single Family Homes, 3+ Floors	800,000					130	All have only 3 floors
Apartment Buildings, 2-4 Units, 1-2 Floors	1,950,000	67%	12%	18%	4%	304	All have 4 units; building count split evenly between 1 and 2 floors
Apartment Buildings, 5+ Units, 1-2 Floors	820,000	66%	16%	16%	2%	612	All have 10 units; building count split evenly between 1 and 2 floors
Apartment Buildings, 5+ Units, 3-4 Floors	300,000					470	All have 20 units; building count split evenly between 3 and 4 floors
Apartment Buildings, 5+ Units, 5-10 Floors	32,000					532	All have 50 units; building count split evenly between 5 through 10 floors
Apartment Buildings, 5+ Units, 11-20 Floors	600					259	All have 100 units; building count split evenly between 11 through 20 floors

* Note that the building characteristics for towns are not used in this study, but they are shown here for completeness. To convert m^2 to ft^2, divide by about 0.093.

Table 6. Estimated Footprint of the Average Residence Building in each Location Type*

	Cities	Suburbs	Towns	Rural Areas
Avg Residence Building Footprint (m^2)	203	196	198	193

*Note that towns are not used in this study, but they are shown here for completeness. To convert m^2 to ft^2, divide by about 0.093.

For each study site, the land area covered by yards was estimated by subtracting the land area covered by residential buildings per city block from the area of each city block. The area of each city block was calculated by multiplying together the length and width of the city block. Yards were linked to the land-use type "Urban/Recreational Grasses", which is defined as "Vegetation (primarily grasses) planted in developed settings for recreation, erosion control, or aesthetic purposes. Examples include parks, lawns, golf courses, airport grasses, and industrial site grasses" (USGS, 2010). This land use designation is shown in Table 7, which also shows that the cumulative yard area per city block ranges from about 5,749 m^2 at the urban study site to about 19,040 m^2 at the suburban study site.

For each study site, the land area covered by roads per city block was calculated by allocating to the block half the width of each road bordering the block. Roads were linked to the land use type "Commercial/Industrial/Transportation", which is defined as "Includes infrastructure (e.g. roads, railroads, etc.) and all highly developed areas not classified as High Intensity Residential" (USGS, 2010). This land use designation is shown in Table 7, which also shows that the cumulative road area per city block ranges from about 1,744 m^2 at the urban study site to about 2,504 m^2 at the suburban study site.

These land use distributions are combined with season and rainfall information to determine the monthly values of the three surface characteristics. The climate information needed to determine seasons and rainfall quantities is described below.

Meteorology Parameters

All three scenario locations use meteorological data from Boston Logan International Airport. The wind direction was predominantly from the west-northwest, as shown in Figure 8. Exact windfields will vary throughout the country due to climatology and microclimatic factors. However, the predominant wind direction in this dataset is consistent with general mid-latitude westerly wind flow. The windspeed (on average 10 knots or 11 mph) is also generally reflective of typical northeastern and western-midwestern average windspeeds where the proxy cities are located (see http://hurricane.ncdc.noaa.gov/climaps/wnd60a13.pdf).

AERMET requires hourly surface data and twice-daily upper-air data. The hourly surface data for Boston Logan International Airport (Weather-Bureau-Army-Navy (WBAN) identifier 14739) were obtained from National Climatic Data Center (NCDC) and are in Integrated Surface Data Tape Data-3505 format (NCDC ISD, 2010). These surface hourly data were formatted as necessary for use in AERMET, and only the official end-of-hour observations were used. The closest upper-air station to Boston Logan International Airport is located in Chatham, MA (WBAN identifier 14684). The upper-air data for Chatham were obtained from the National Oceanic and Atmospheric Administration/Earth System Research Laboratory Radiosonde Database Access (NOAA ESRL, 2010). The upper-air data are in AERMET-friendly Forecast Systems Laboratory format, and only the official 00 Coordinated Universal Time (UTC) and 12 UTC observations at mandatory and significant atmospheric levels were used. In order to model air concentrations and deposition using the most recent 12-month meteorological data, the surface and upper-air data were obtained for August 2009 through July 2010.

AERMET also requires three important surface characteristics – surface roughness length, albedo, and Bowen ratio. The values of the surface characteristics for a given land use type can vary by season, so the user must define the seasons of the study sites.

Because Boston Logan International Airport is being used as the meteorological proxy for this study, the climatology of the airport area was analyzed in order to define which month is part of which season.

First, winter must be defined as snowy or not snowy, where snowy is defined as experiencing continuous snow cover for at least one month per year. As described in U.S. EPA (2009f), the shapefiles from the NCDC Climate Maps of the United States database (NCDC, 2005a) were used to analyze typical snow cover at any location in the lower 48 U.S. states. By this analysis, the Boston Logan International Airport location met this definition of snowy.

Table 7. The Land Use Characteristics of Each Study Site

	Urban Study Site	Rural Study Site	Suburban Study Site
Area of City Block, Including Half of Roads on Every Side (m^2)	10,744	15,129	23,504
Cumulative Area of Residence Buildings per City Block (m^2)	3,251	1,160	1,960
% of Area of City Block that is Comprised of Residence Buildings	30%	8%	8%
Land Use Type for Residence Buildings	High Intensity Residential	Low Intensity Residential	Low Intensity Residential
Cumulative Area of Yards per City Block (m^2)	5,749	12,065	19,040
% of Area of City Block that is Comprised of Yards	54%	80%	81%
Land Use Type for Yards	Urban/ Recreational Grasses	Urban/ Recreational Grasses	Urban/ Recreational Grasses
Cumulative Area of Roads per City Block, With Half of Roads Included on Every Side (m^2)	1,744	1,904	2,504
% of Area of City Block that is Comprised of Roads	16%	13%	11%
Land Use Type for Roads*	Commercial/Industrial /Transportation (non-airport)	Commercial/Industrial /Transportation (non-airport)	Commercial/Industrial /Transportation (non-airport)

* The land use types correspond to those contained in the 1992 NLCD (USGS, 2010).

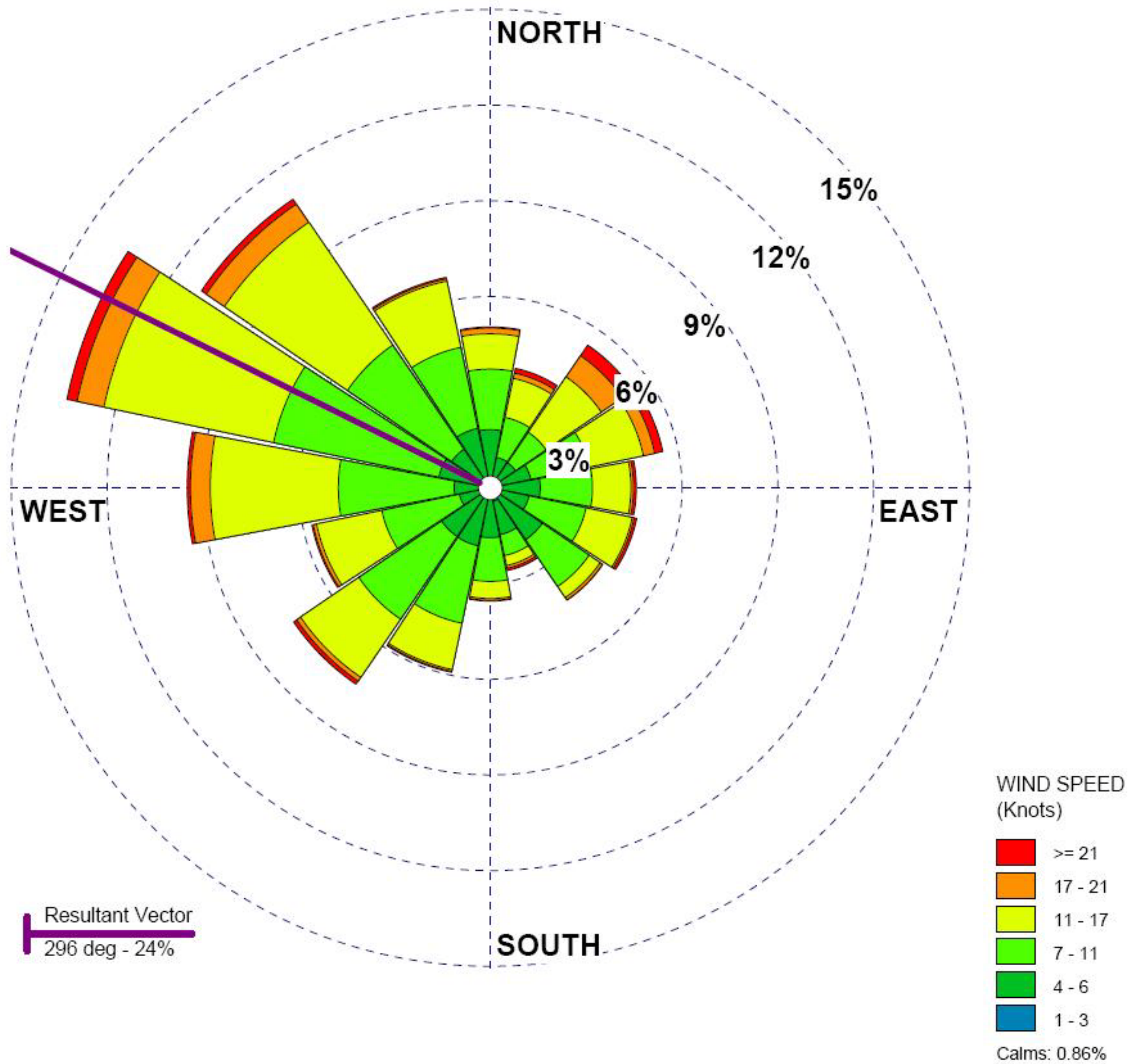

Figure 8. Wind Rose for Boston Logan Airport Meteorological Station

Second, each month must be assigned to a season. The same procedures used in *Risk and Exposure Assessment to Support the Review of the SO_2 Primary National Ambient Air Quality Standards* (U.S. EPA, 2009f) to determine seasons for the lower 48 U.S. states were used in this study. As with defining continuous snow cover, the procedures for defining seasons relied on data from NCDC (2005a). Based on these criteria, winter at the Boston Logan International Airport location was defined as December through February, spring was defined as March through May, summer was defined as June through August, and autumn was defined as September through November.

Finally, the AERSURFACE look-up tables require information as to whether the location was experiencing above average, below average, or average precipitation on a monthly basis. To determine the precipitation category, the AERSURFACE guidance recommends comparing the period of record of the meteorology data used in the modeling to the 30-year period of record for the same location and selecting above average if the modeling period is in the upper 30th percentile of the 30-year record, below average if in the lower 30th percentile, and average if otherwise. AERSURFACE applies this precipitation designation to the whole period of modeling. For the August 2009 through July 2010 period of modeling for this study, the 12-month total precipitation was 53.44 inches (135.7 cm) at the Boston Logan International Airport, which is 26% above the 1971-2000 Climate Normals annual precipitation amount of 42.53 inches (108 cm) (NCDC, 2005b).

Table 8. Comparison of Monthly Precipitation to Average Conditions to Determine Precipitation Category

	Jan	Feb	Mar	Apr	May	Jun	Jul	Aug	Sep	Oct	Nov	Dec	Ann
August 2009-July 2010 Monthly Precipitation Amount (cm)	6.10	7.59	39.98	4.65	8.51	11.56	7.24	8.41	8.00	14.27	9.32	10.11	135.74
1971-2000 70th Percentile Monthly Precipitation Amount (cm)	12.30	8.98	10.89	11.16	9.13	7.75	10.15	11.08	11.66	10.87	13.03	12.88	115.61
1971-2000 30th Percentile Monthly Precipitation Amount (cm)	6.35	6.33	6.36	6.05	5.60	3.79	5.46	4.06	3.98	7.48	6.26	5.69	96.09
"Wetness" Category for 2009-2010 Data (used for AERSURFACE)	DRY	AVG	WET	DRY	AVG	WET	AVG	AVG	AVG	WET	AVG	AVG	WET
Season	Winter (snowy)	Winter (snowy)	Spring	Spring	Spring	Sum-mer	Sum-mer	Sum-mer	Autumn	Autumn	Autumn	Winter (snowy)	--

However, individual months of the period of modeling range from 49% drier than normal to over 300% wetter than normal. Because this study will calculate monthly values of surface roughness length, albedo, and Bowen ratio, and because of these large monthly variances in precipitation, it is useful to categorize the precipitation amounts on a monthly basis. Monthly precipitation categories were also used in the NO_2 NAAQS risk analysis (U.S. EPA, 2008c), where AERSURFACE was run three times (once per precipitation setting), and the monthly values of the three surface characteristics using the three precipitation settings were merged according to monthly precipitation.

Monthly precipitation amounts from NWS (2005) were compared against the August 2009 through July 2010 monthly precipitation amounts. As shown in Table 8, two of the 2009-2010 months experienced precipitation amounts that were less than their respective

30th percentile 1971-2000 values. Three of the months experienced precipitation amounts that were greater than their respective 70th percentile 1971-2000 values. The other seven months experienced precipitation amounts that were within their respective 30th and 70th percentile values.

The culmination of the land use and climate characteristics is shown in Table 9. It shows the values of the three surface characteristics (albedo, Bowen ratio, and surface roughness length) for each month and for each scenario location type (urban, rural, and suburban). For each location type, these values were determined by averaging together the values of each surface characteristic for each land use type specific to the location. The averaging is weighted by the area of each land use type per city block. The surface characteristic value look-up tables are provided in Appendix A of the AERSURFACE User's Guide (U.S. EPA, 2008a). The areas of each land use type per study site are shown in Table 7, and the season and "wetness" category assigned to each month are shown in Table 8.

Table 9. Model Values of Albedo, Bowen Ratio, and Surface Roughness Length for each of the Three Study Scenario Types *

Month	Season	"Wetness" Category	Albedo			Bowen Ratio			Surface Roughness Length (m)		
			Urban	Rural	Suburban	Urban	Rural	Suburban	Urban	Rural	Suburban
Jan	Winter	Dry	0.48	0.56	0.56	0.50	0.50	0.50	0.44	0.14	0.13
Feb	Winter	Avg	0.48	0.56	0.56	0.50	0.50	0.50	0.44	0.14	0.13
Mar	Spring	Wet	0.48	0.15	0.15	0.57	0.33	0.32	0.44	0.15	0.14
Apr	Spring	Dry	0.48	0.15	0.15	1.93	1.33	1.30	0.44	0.15	0.14
May	Spring	Avg	0.48	0.15	0.15	0.86	0.49	0.47	0.44	0.15	0.14
Jun	Summer	Wet	0.16	0.15	0.15	0.63	0.41	0.40	0.44	0.16	0.15
Jul	Summer	Avg	0.16	0.15	0.15	0.96	0.65	0.63	0.44	0.16	0.15
Aug	Summer	Avg	0.16	0.15	0.15	0.96	0.65	0.63	0.44	0.16	0.15
Sep	Autumn	Avg	0.16	0.15	0.15	1.07	0.82	0.81	0.44	0.15	0.14
Oct	Autumn	Wet	0.16	0.15	0.15	0.68	0.49	0.48	0.44	0.15	0.14
Nov	Autumn	Avg	0.16	0.15	0.15	1.07	0.82	0.81	0.44	0.15	0.14
Dec	Winter	Avg	0.48	0.56	0.56	0.50	0.50	0.50	0.44	0.14	0.13

* These values were derived from the tables in Appendix A of the AERSURFACE User's Guide (U.S. EPA, 2008a), along with the "wetness" and season designations shown in Table 5 and the land use characteristics shown in Table 7.

Release Height and Dimensions

AERMOD requires the following parameters to be assigned for each source: Emission Rate (Aermis), Release height (Relhgt), width of roadway (Xinit) and initial vertical dimension (Szinit) (U.S. EPA 2004). Average release heights and initial vertical dimensions for light-duty and heavy duty vehicles are presented in "Transportation Conformity Guidance for Quantitative Hot-spot Analysis in $PM_{2.5}$ and PM_{10} Nonattainment and Maintenance Areas" (U.S. EPA 2010d). Table 10 below lists default values by vehicle type. The lead dust is assumed to be lifted from the ground due to turbulence from passing vehicles, and this turbulence leads to the further emission of lead dust from the roadway to the air. Because the turbulence extends approximately over the height of the vehicle creating it, the release heights correspond roughly to vehicle heights. Site specific vehicle type distributions were obtained from MOVES (U.S. EPA, 2009e) and a class-weighted average was applied to get site-specific release height and initial vertical dimension values for each scenario (see Table 11). This method is consistent with U.S. EPA (2010d) recommendations.

Table 10. Default Release Height and Initial Vertical Dimension for AERMOD modeling

Vehicle Type	Release Height (Relhgt)	Initial Vertical Dimension (Szinit)
Light-duty	1.3 m	1.2 m
Heavy-duty	3.4 m	3.2 m

Table 11. Calculation of Release Height and Sigma Z for Scenarios A-E

Location	Light-duty vehicle distribution*	Heavy-duty vehicle distribution*	Release Height (m)	Sigma Z (m)
Scenario A,B (urban)	85.3%	14.7%	= (1.3×0.853) + (3.4×0.147) = 1.61 m	= (1.2×0.853) + (3.2×0.147) = 1.49 m
Scenario C,D (downtown rural)	81.8%	18.2%	= (1.3×0.818) + (3.4×0.182) = 1.68 m	= (1.2×0.818) + (3.2×0.182) = 1.56 m
Scenario E (suburban)	82.8%	17.2%	= (1.3×0.828) + (3.4×0.172) = 1.66 m	= (1.2×0.828) + (3.2×0.172) = 1.54 m

* Calculated from MOVES; “Heavy Duty” is the sum of vehicle population for “Combination Long-Haul Truck”, “Combination Short-Haul Truck”, “Intercity Bus”, “Light Commercial Truck”, “Motor Home”, “School Bus”, “Single Unit Long-Haul Truck”, “Single Unit Short-Haul truck”, and “Transit Bus” divided by the total population; “Light-Duty” is the sum of vehicle population for “Motorcycle,” “Passenger Car” and “Passenger Truck” divided by the total vehicle population.

Mass Fractions and Particle Diameters

A requirement of AERMOD deposition Method 2 is the fraction of fine particulate matter (< 2.5 μm) in total particulate matter for the road-dust which will be modeled and the mass-median particle diameter (MMAD). Samara and Voutsa (2005) reported size distributions of roadside particulate matter and the MMAD near a roadway in Thessaloniki, Greece. The average mass-median particle diameter was 0.85 ± 0.71 μm. Samara and Voutsa (2005) reported average concentrations of roadway dust for the following size categories:

Average concentration of PM by size (N=32), in μg/m^3:

< 0.8 μm: 54.2 ± 22.2

0.8 – 1.3 μm: 6.59 ± 6.79

1.3 – 2.7 µm: 5.68 ± 3.37

2.7 – 6.7 µm: 16.7 ± 9.34

> 6.7 µm: 23.0 ± 14.3

To calculate the fraction of fine particulate matter, the average concentrations in size categories below 2.7 µm were summed and divided by the sum of concentrations in all categories. This results in a fraction of fine particulate matter of 0.626 for road dust. Implicit in this calculation is the assumption that the lead-containing dust from wheel weights will follow the same size distribution as roadway dust of other sources, although this assumption cannot be verified in the literature.

4.3 Yard Soil Module

The yard soil module predicts the yard lead concentrations at the different receptor yards using the AERMOD wet and dry deposition values and other input values, as depicted in Figure 9. Section 4.3.1 describes the selected assessment method and Section 4.3.2 describes how each parameter value was selected.

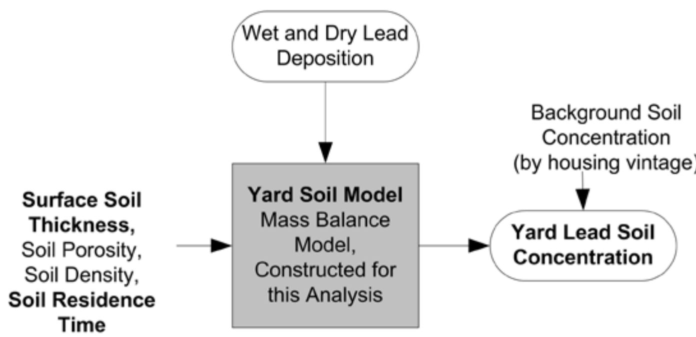

Figure 9. Flowchart Showing the Approach for the Yard Soil Module

4.3.1 Assessment Method Selected

Because AERMOD is strictly an air dispersion model and does not contain a soil module, another model must be found or built which estimates the soil concentrations based on the lead which is deposited from the air and any removal mechanisms. A multimedia model such as TRIM.FaTE models removal processes from colloidal transport from the surface soil compartment to deeper soil layers, lateral runoff, and lateral erosion. However, erosion and runoff will be dependent on the meteorology and the topography of the modeling domain, and uncertainties in each will introduce uncertainties in the results. Given the large uncertainties in the emission data, a simpler modeling approach was favored.

Thus, to estimate the contribution to the yard soil concentration from the wheel weight lead emission, a simple steady-state vertical mass balance model was constructed and parameterized.

Let:

M = the mass of lead in the soil (in μg)

C = the concentration of lead in the soil (in μg/g)

D = the deposition rate of lead into the soil (in $\mu g/m^2$/year)

τ = the residence time of lead in soil (in years)

ρ = the density of the soil (g/m^3)

ϕ = the porosity of the soil (fraction)

A = the area of the yard (m^2)

d= the depth of the top soil layer (m)

Mass balance considerations dictate that:

> The change in the mass of lead in the soil equals the deposition input from above less the loss due to vertical colloidal transport.

Using the symbols defined above, this mass balance equation may be expressed mathematically in terms of the following differential equation:

$$\frac{dM}{dt} = D \times A - \frac{M}{\tau} \qquad (5)$$

This equation assumes that the colloidal transport can be captured by first order removal with a rate constant equal to $1/\tau$ (which is equivalent to the residence time). At steady state, the mass of lead in the soil is not changing, so

$$\frac{M}{\tau} = D \times A \qquad (6)$$

The mass of lead in the soil can be converted to concentration in units of mass of lead per mass of soil by using the soil density, porosity, and soil thickness,

$$C = \frac{D \times \tau}{d \times \rho \times (1-\phi)} \qquad (7)$$

Thus, given the total deposition of lead in the yard from the AERMOD model, the residence time in the soil, the soil depth, the soil density, and the porosity, the lead concentration due to wheel weights can be calculated using equation (7). Then, the wheel weights contribution can be subtracted from the total soil lead concentration to estimate a "no wheel weights" soil concentration.

The assessment framework for the near-roadway residence includes resuspension of road dust into the air and the subsequent dispersion and deposition of this lead-containing dust into nearby yards. However, the approach does not include the resuspension of contaminated yard soil into the air. In order to include this process, a full multi-media model that simultaneously models both air and soil processes would have to be used;

however, these models tend to have less sophisticated dispersion algorithms than the air-only AERMOD model.

To determine the possible uncertainty associated with excluding yard soil resuspension, a literature search was conducted. In general, the papers suggest that resuspension of contaminated soil can be a large contributor to ambient air concentrations. Harris and Davidson (2009) employ a mass balance model to conclude that sources of lead due to the resuspension of contaminated soil/dust are a factor of ten higher than direct sources of lead in the South Coast Air Basin in California. They cite the main contributor of lead in the soil to be from historical deposition in the era of leaded gasoline, and the current sources due to resuspension include both yard soil and roadway soil. Sabin et al. (2006), however, found that much of the airborne lead in Los Angeles was due to resuspension from roadways, and concentrations of lead in air returned to near-background levels within 10 to 150 m of the roadway. Hosiokangas et al. (2004) also found that roadways were a major contributor to airborne lead levels (27%) in Finland, and the windspeed tended to be the major determinant of how much lead was resuspended. These papers suggest that resuspension of contaminated soil/dust is a major contributor to airborne lead, but much of this resuspension occurs on roadways where car turbulence creates an effective mechanism for suspending the dust. Thus, excluding yard resuspension will tend to under-predict the yard air lead concentrations; however, the dominant source to a yard next to a roadway is likely the resuspended roadway lead rather than lead resuspended from the yard itself. The exclusion of yard resuspension remains a recognized limitation of the modeling approach.

4.3.2 Parameter Selection

Surface Soil Thickness

The thickness of the surface soil layer assumed in TRIM.FaTE model simulations performed for EPA OAQPS ranges from 1 cm for non-agricultural soils to 20 cm for tilled agricultural soils (U.S. EPA, 2009c). Although yard soils are not expected to be tilled, they may be mowed, raked, landscaped, or used for gardening. Due to the wide variability, a yard surface soil layer thickness of 1 cm was assumed. Because this parameter has a wide range in the literature, it is considered highly uncertain. An additional uncertainty analysis using an alternative thickness of 10 cm is presented in Section 4.9.

Soil Porosity and Density

The soil particle density of 2,600 $kg/m^3$was taken from the CalTOX model (McKone and Enoch, 2002). CalTOX is a model developed by funding from the U.S. EPA to model the environmental fate of chemical in air, soil and water and has been applied to a number of chemical risk assessments. In addition, the soil porosity was set to the CalTOX value of 20% or 0.2.

Soil Residence Time

A literature search was conducted to estimate the residence time of lead in surface soil. The following studies were reviewed: Tyler (1978), Miller and Friedland (1994), Erel (1998), U.S. EPA (2001), Kaste et al. (2003), Semali et al. (2004), Kaste et al. (2005), Klaminder et al. (2006a), Klaminder et al. (2006b), and Mireztky and Fernandez-Cirelli (2007), as shown in Table 12.

There were a number of variations in each of the studies reviewed. Studies were conducted in different areas of the world, including the Northeastern United States, Israel, Sweden, and France. Studies derived the residence time using a number of different methods, including experimental measurement of lead through soil, mass-balanced source models, tracer isotope tracking within soil, or chronosequencing lead in soil gradients. In addition, results were presented in numerous formats including residence times, response times, half lives, and 10% removal times. All half-life and 10% removal calculations were converted to response time, and calculations were made to ensure all definitions in the papers of residence time and response time were equivalent to each other.

Table 12. Lead in Soil Residence Time Literature Search Results

Paper	Year	Reported Time (yrs)	Residence Time (yrs)	Location
Tyler (1978)	1978	700-900 (10% removal)	6650-8550	Forest in Sweden
Miller and Friedland (1994)	1994	17-77 (response)	17-77	Northeast US
Erel (1998)	1998	100-200 (residence)	100-200	Israel, farmland and forest
U.S. EPA (2001)	2001	1000 (half life)	1442	Unknown
Kaste et al. (2003)	2003	60-150 (response)	60-150	Northeast US
Semali et al. (2004)	2004	700 (half life)	1000	France
Kaste et al. (2005)	2005	50-150 (response)	50-150	Northeast US
Klaminder et al. (2006a)	2006	150 (residence)	150	Forest in Sweden
Klaminder et al. (2006b)	2006	50-250 (residence)	50-250	Forest in Sweden
Mireztky and Fernandez-Cirelli (2007)	2007	740-5900 (half life)	1070-8500	Unknown

A number of factors affect the residence time of lead in the soil. The carbon flux within the soil layer is closely correlated with the residence time of lead. In newer growth forests, residence times are smaller than older growth forests. There is greater turnover of carbon in these newer growth forests. Older growth forests may have a higher organic carbon content in the upper layers or soil, but it may be broken down more slowly (Klaminder et al., 2006b). In addition, warmer climates may have quicker turnover of carbon and thus shorter lead residence times (Miller and Friedland, 1994).

Overall, the values reported in the studies vary over a wide range. For the yard soil module, a value of 1,000 years was selected. This value is in the middle of the range of literature values for the residence time. Because the range of values in the literature is so large, this variable is considered to be highly uncertain. In order to determine the effect of varying this parameter to a lower value, an uncertainty analysis using a residence time of 150 years is presented in Section 4.9.

Total Soil Concentration

Total home yard lead soil concentrations were determined for the model scenarios using proxy locations for each type, as shown in Table 13. For the urban location, a high soil concentration was used. The value was taken from a study of the concentrations in yards in Dorchester, MA (Hynes et al., 2001). The selected value represents the arithmetic mean of lead in surface soil in the North Dorchester section of Boston.

For the rural location, both high and low soil concentration areas are modeled. For the high soil concentration yard, values from a study measuring soil concentrations in residential Minnesota were used (Schmitt et al., 1988). The value represents the maximum value for the front yard lead concentrations for the “outstate” classification. For the low soil concentration area, values from a study measuring lead concentration in rural topsoil in South Carolina were used (Aelion et al., 2008). The value represents the mean lead concentration in the less contaminated strip of land from the study (strip 1).

For the suburban location, a low soil concentration area is modeled as it was assumed that this would be a post 1980’s housing development. The Schmitt et al. study mentioned above for rural locations was used, and the selected value represents the median front yard lead concentrations for the "outstate" classification.

Table 13. Total Home Yard Lead Soil Concentration

Urban, High Soil Concentration (Scenarios A and B)	Rural, High Soil Concentration (Scenario C)	Rural, Low Soil Concentration (Scenario D)	Suburban, Low Soil Concentration (Scenario E)
1463 μg/g	656 μg/g	12 μg/g	37 μg/g

4.4 Indoor Air/Dust Module

The indoor air/dust module estimates the indoor air lead concentration from the ambient concentration using a penetration factor. It also estimates the indoor dust concentration using a regression model, the vintage of the home, and the calculated soil lead concentrations at the home, as depicted in Figure 10. Section 4.4.1 describes the assessment method selected for this module. Section 4.4.2 describes the selection of the input parameter values.

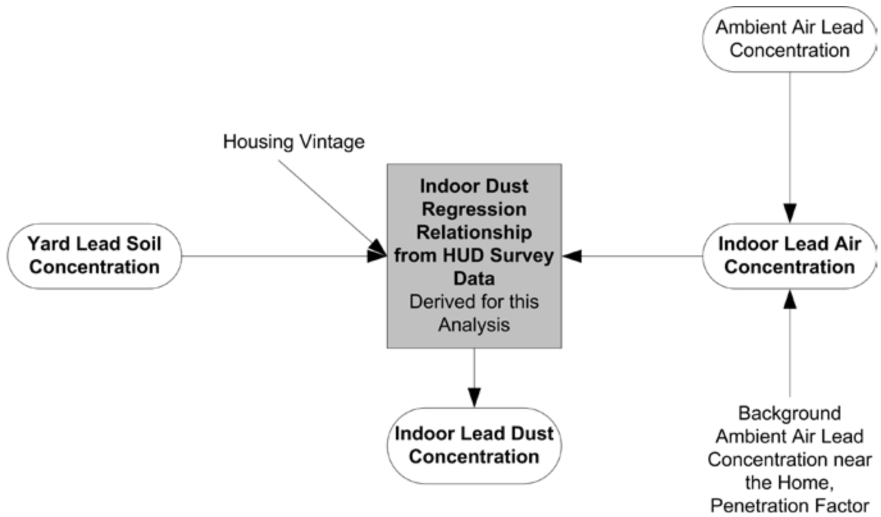

Figure 10. Flowchart Showing the Approach for the Indoor Air/Dust Module

4.4.1 Assessment Method Selected

The concentration of lead in indoor dust inside a home is determined by the outdoor soil concentration tracked into the home, the indoor lead paint concentration in the home, the ambient air concentration, the cleaning frequency, the occupancy level and characteristics, and the nature of non-lead particulate sources in the home. Lead wheel weights will contribute lead mass to the outdoor soil concentration and ambient air concentration, which will in turn affect the indoor lead dust concentration. In addition, different housing vintages in the different scenarios will have different levels of lead in the interior paint.

To fully capture the effect of the ambient air concentration and the soil concentration from wheel weights on the indoor lead levels, a fully physical model would need to be built that parameterizes the fate and transport of air particles and tracked-in soil particles in the home. In addition, because blood lead models generally accept only lead dust concentration (and not lead dust loading), a model would also need to be used to convert lead mass loadings to concentrations. However, in a fully physical model, all of the source and removal terms would include numerous parameters each with their own uncertainties. Given the uncertainties in the wheel weight loss and pulverization rates, a simpler assessment method was favored.

A literature search was conducted to find a dataset that simultaneously measured outdoor or indoor air concentrations, outdoor soil concentrations, and indoor dust concentrations. No such dataset at the national level could be identified. However, the National Survey of Lead-Based Paint in Housing ("HUD Survey Data", U.S. EPA, 1995) provides information on the lead dust concentration determined from particulate collected using

Blue Nozzle vacuum samplers, yard-wide average lead soil concentrations, the maximum observed indoor XRF lead paint concentrations, and the housing vintage for 312 homes. These data were used in this assessment to derive a regression equation relating the total interior dust concentration (including wheel weight sources and all other sources) with the outdoor soil concentration and the paint concentration. The ambient air concentrations were not captured in the survey, so these values could not be included in the regression equation.

Using Statistica®, a multiple linear regression equation was developed relating the indoor dust concentration to the outdoor soil concentration and indoor paint concentration. Both the untransformed and the natural-log-transformed variables were used in order to determine which linear regression captured the largest portion of the observed variance. Statistics from the two different fits are shown below in Table 14. The regression based on the untransformed variables captured little of the total variance and did not indicate significance at the p=0.01 level. Thus, the regression based on the natural-log-transformed variables was selected. This regression has an adjusted R^2 of 0.24, representing modest predictive power and indicating much of the variance is explained by other factors not included in the regression or captured in the survey, such as those mentioned above (ambient air concentration, cleaning frequency, occupancy level, etc.). The equation for the indoor dust concentration in μg/g becomes

$$\text{Dust} = 44.3 \times \text{Soil}^{0.33} \times \text{Paint}^{0.22}$$

where *Soil* is the concentration in the soil in μg/g and *Paint* is the concentration of lead in the interior paint in mg/cm^2. Figure 11 shows the predicted natural log of dust as a function of the observed natural log of dust, where the solid line denotes a 1:1 correspondence.

Paint concentrations are not explicitly considered in the overall wheel weight modeling approach. However, the housing vintage in each scenario has been specified. Thus, the average paint concentration across all homes in the HUD Survey in each specified vintage category was calculated and plugged into the dust equation to create vintage-specific equations, as shown in Table 15 below.

Table 14. Statistics of the Multiple Linear Regression for Dust Concentration

	R	R^2	Adjusted R^2	P level	Standard Error
Untransformed	0.047	0.0022	--	< 0.72	2961.8
Natural-log-transformed	0.5	0.25	0.24	< 1e-10	1.08

Figure 11. Predicted ln(Dust) as a Function of the Observed ln(Dust)

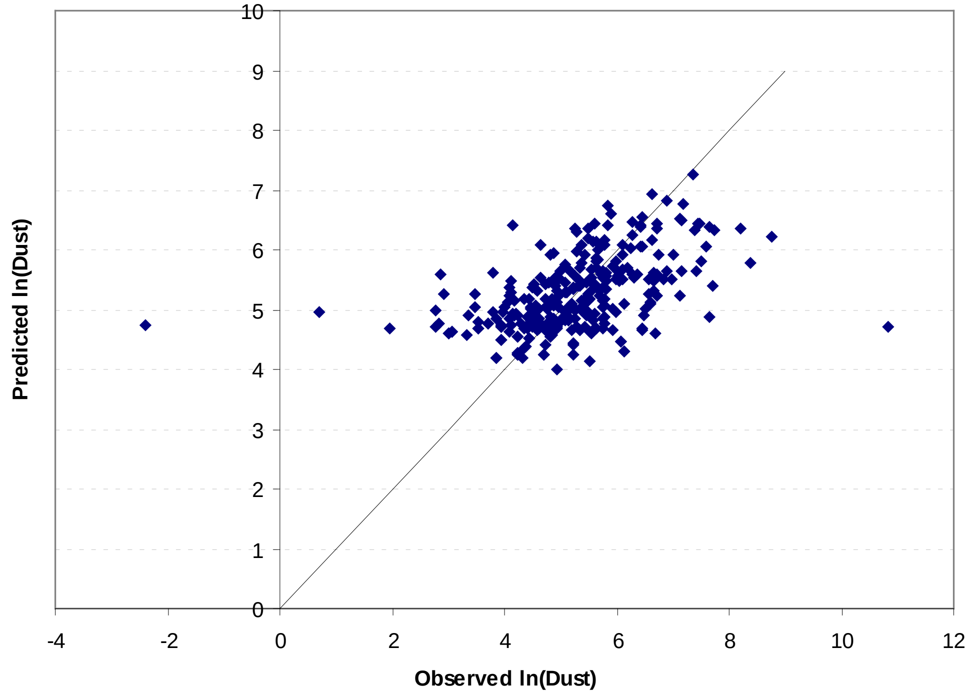

Table 15. Dust Regression Equation By Housing Vintage

	Pre 1940 Vintage (Scenarios A and C)	Post 1980 Vintage (Scenarios B, D, and E)
Average XRF Paint Concentration (mg/cm^2)	3.69	0.519
Dust Equation	$\text{Dust} = 59.0 \times \text{Soil}^{\ 0.33}$	$\text{Dust} = 38.3 \times \text{Soil}^{\ 0.33}$

4.4.2 Parameter Selection

Total Ambient Air Concentration

The total ambient air concentration was calculated using air monitoring information from the EPA's Air Quality System (AQS; U.S. EPA, 2010a) DataMart database. Average annual concentrations from all monitoring locations in the AQS system measuring lead total suspended particulate (TSP) at standard temperature and pressure (STP), or parameter ID 12128. Data from 2008 were used, since in 2009 monitors began using updated reporting methods due to the most recent lead NAAQS rules; however, because different monitors used different reporting methods, the statistical strength of averaging for any one reporting type was greatly diminished.

The AQS database includes a field named "Monitoring Objective" that specifies the reason that a monitor was placed in each location. Monitors labeled "source oriented", "quality assurance" (duplicate monitors at the same site, which may bias results), or "Unknown" were removed from the analysis, as it is likely that the results from these sites will bias total ambient air concentrations. In addition, numerous monitors were located in the town of Herculaneum, Missouri which is the site of the largest lead smelter in the United States. All sites located in Herculaneum were also removed, regardless of the stated monitoring objective.

Monitoring stations were assigned to rural, suburban, or urban locations in AQS using the "Location" field. If the location was unknown, the latitude and longitude was viewed in Google Earth® and an assignment was made by professional judgment. Only locations with residential and commercial land use types were included.

The remaining monitors' annual average concentrations in $\mu g/m^3$ for each station type (rural, suburban, or urban) were used to give estimates of the average, standard deviation, and median ambient air concentrations in each location, as shown in Table 16. The average concentrations were selected for use in the modeling framework.

Table 16. Ambient Air Concentrations from the AQS Monitoring Network

Description	N	Average ($\mu g/m^3$)	Standard Deviation ($\mu g/m^3$)	Median ($\mu g/m^3$)
Urban and City Center	31	0.025	0.054	0.0075
Rural	8	0.011	0.006	0.0130
Suburban	39	0.014	0.022	0.0067

Penetration Fraction of Ambient Air Into Home

The penetration fraction captures the ratio of the indoor concentration from outside sources to the ambient (outdoor) concentration. The penetration fraction was set equal to 1.0, taken from Thatcher and Layton (1995). The paper reported penetration for lead-containing particles in a home in California, and the penetration fraction was near one for all size classes. Thus, the indoor air concentrations used in the blood lead modeling are set equal to the outdoor air concentrations.

4.5 Blood Lead Module

The blood lead module uses the lead soil, air, and dust concentrations calculated above as inputs, as depicted in Figure 12. In addition, water and dietary concentrations as well as other exposure inputs are specified. The output of the module is the average blood lead in the child or adult living near the roadway. Section 4.5.1 describes the assessment methods selected for children and adults. Section 4.5.2 describes how the parameters for each model were selected.

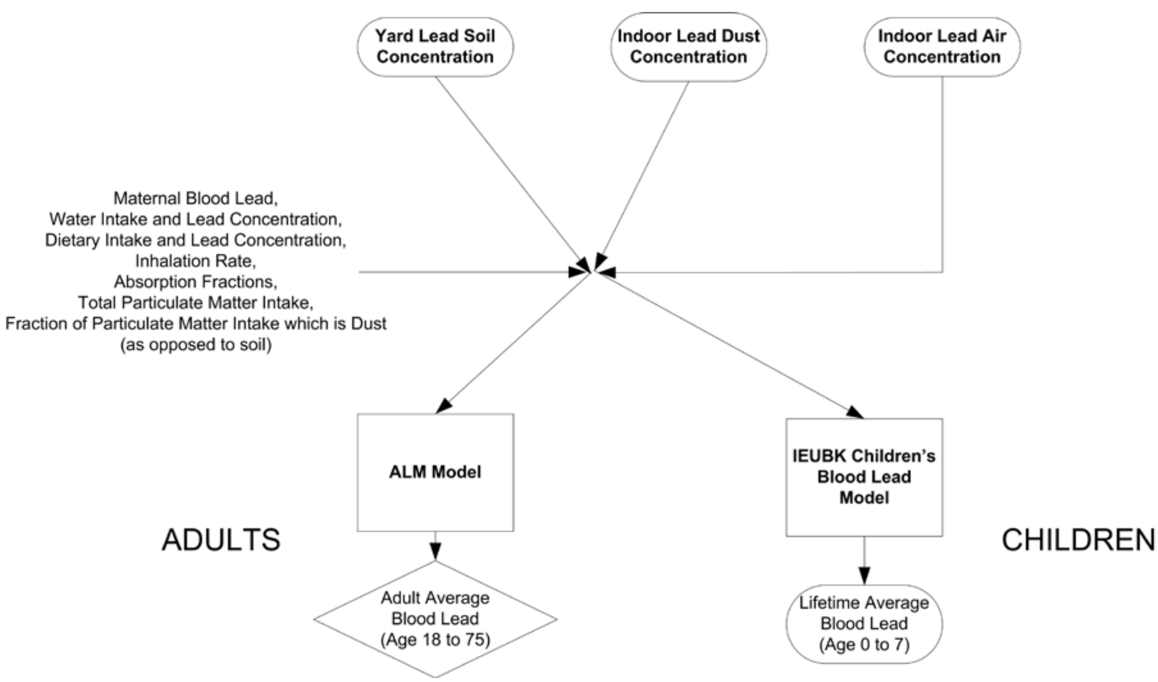

Figure 12. Flowchart Showing the Approach for the Blood Lead Module

4.5.1 Assessment Method Selected

Several models are available to estimate the blood lead levels for children and adults. The relative merits of each are discussed in recent EPA publications (e.g., U.S. EPA, 2007a and U.S. EPA, 2007b). The Integrated Exposure Uptake Biokinetic (IEUBK) model (U.S. EPA, 2010c) is a model for children from birth up to age seven. It has undergone extensive evaluation and validation by EPA scientists and outside reviewers (Mickle, 1998). Another model, the Leggett model (Leggett, 1992), can be used for children or adults and allows exposure concentrations and biokinetic parameters to change from birth to age seventy-five and above. It tends to predict childhood exposures which are two to three times higher than the IEUBK model (U.S. EPA, 2007a). IEUBK has been compared with measurements in NHANES and tends to predict blood lead values that are more consistent with population means than the Leggett model (U.S. EPA, 2007a). However, the Leggett model is better at capturing acute exposures to high lead levels in the exposure media, since biokinetic parameters and exposure values can vary on timescales shorter than a month in the Leggett model but not the IEUBK model.

Because the current exposures are assumed to remain constant throughout the life of the child (as opposed to a very short duration “spike” of exposure during a renovation activity) and because the IEUBK model tends to compare more favorably with NHANES data for children, the IEUBK model was selected to estimate children’s exposure to lead in wheel weights. The model was run for each year age 0 to7 and then a lifetime-average blood lead was calculated.

The IEUBK model, which can estimate blood lead levels only in children up to age 84 months, was not used to predict adult blood lead levels. As an alternative, EPA’s Adult Lead Methodology (ALM) (U.S. EPA, 1996; U.S. EPA, 2003), which uses a linear “biokinetic slope factor” (BKSF) to estimate lead dose from soil exposure, was adapted as described below. For comparison purposes, the Leggett model was also used to estimate exposures for adults and the results are shown in the appendix.

EPA originally developed the ALM (U.S. EPA, 1996) to estimate blood-lead impacts of exposures to lead-contaminated soil near “Superfund” sites. The approach was subsequently modified and refined, with a focus on evaluating blood lead impacts in women of childbearing age (U.S. EPA 2003) and predicting the proportion of exposed women and fetuses with blood-lead levels above levels of 10 μg/dL. The structure of the ALM is simple: estimates of steady-state (long-term) blood-lead concentrations are estimated as a linear function of soil exposures. Exposure concentrations are used to estimate time-averaged blood-lead uptake (absorbed dose) based on exposure factors (exposure frequency, soil ingestion rate, gastrointestinal absorption fraction) that are judged to be typical of the exposed population. In the simplest form of the ALM, the predicted central tendency blood-concentration is given by:

$$PbB_{adult} = PbB_{adult,0} + BKSF \times UP_S$$

$$UP_S = \frac{PbS * IR_S * AF_S * EF_S}{AT}$$

where:

$PbB_{adult,\,0}$= typical central tendency blood lead concentration in the absence of soil exposures (μg/dL)

BKSF = biokinetic slope factor (μg/dL per μg/day lead uptake)

UP_S = total soil lead uptake (μg/day)

PbS = soil lead concentration (μg/g)
IR_S = average soil ingestion rate (g/day)
AF_S = gastrointestinal absorption fraction for lead in soil
EF_S = exposure frequency (days/year)
AT = averaging time (365 days/year for chronic exposures)

In order to adapt the model to apply to wheel weight exposures, the total soil uptake in the model was recharacterized as uptake from ingestion of both lead soil and dust. The biokinetic slope factor can then be applied to the total particulate ingestion rather than just the soil particulate ingestion. The dust lead concentration, the total soil and dust ingestion, and the fraction of total soil and dust ingestion derived from soil were added to the equation in the following way:

$$PbB_{adult} = PbB_{adult,0} + BKSF \times UP_{S+D}$$

$$UP_{S+D} = (PbS \times W_S + PbD \times (1 - W_S)) \times IR_{S+D} \times AF_{S+D} \times EF_{S+D} \times AT$$

where:

$PbB_{adult,\,0}$ = typical central tendency blood lead concentration in the absence of soil and dust exposures (μg/dL)
$BKSF$ = biokinetic slope factor (μg/dL per μg/day lead uptake)
UP_{S+D} = total soil and dust lead uptake (μg/day)
PbS = soil lead concentration (μg/g)
PbD = dust lead concentration (μg/g)
W_S = weighting factor indicating fraction of soil and dust ingestion from soil
IR_{S+D} = average soil and dust ingestion rate (g/day)
AF_{S+D} = gastrointestinal absorption fraction for lead in soil and dust
EF_{S+D} = exposure frequency (days/year), set equal to 365
AT = averaging time (365 days/year for chronic exposures)

By adapting the model in this way, the dust and soil contributions to blood lead from lead wheel weights can be explicitly estimated. However, the air contribution of wheel weight lead to blood lead is not explicitly included. Because the ALM was specifically developed for Superfund applications and exposure due to particulate ingestion, the model was not adapted to include inhalation exposure. However, the uptakes from inhalation exposures to lead in wheel weights are a small proportion of the total uptake (see Section 4.7 for a discussion in children; similar conclusions apply for adults). Small contributions from total inhalation uptakes are included as part of the $PbB_{adult,0}$ term.

4.5.2 Parameter Selection

IEUBK Parameters

IEUBK requires a number of inputs aside from the air, soil, and dust lead concentrations. Table 17 shows the inputs and the proposed values for each. As a starting point, the values were set to those used in the exposure assessment supporting the current lead NAAQS level (U.S. EPA, 2007a) and in the exposure assessment supporting the Lead Renovation, Repair, and Painting rule, or "LRRP" rule (U.S. EPA, 2007b). Then, where possible, values were updated with data from more recently published literature. These included water lead concentration, lead absorption fractions, dietary lead intake, and the fraction of ingested soil and dust from soil.

In 2008, the U.S. EPA published a new edition of its Child-Specific Exposure Factors Handbook, from which updated mean values for total indoor/outdoor dust ingestion, water consumption, and ventilation rate were derived (U.S. EPA, 2008b). Where ages were expressed as a range in that report, rates for intermediate ages were interpolated using linear trendlines.

The IEUBK value for maternal blood lead level was updated using data from the most recent NHANES survey. These data from 2007 and 2008 reveal that the GM blood lead level among women aged 18 through 45 is 0.847 µg/dL. This was computed using the NHANES laboratory sample data and included nationally-representative sample weights (CDC, 2009). This value is somewhat lower than the adult predictions of blood lead for women living near the roadway presented in section 4.8. However, the maternal blood lead does not play a large role in estimating the child's lifetime-average blood lead in the IEUBK model. When the higher values presented in Table 23 are used for the maternal blood lead for each scenario, the lifetime average blood lead values only change by 2% or lower.

Table 17. IEUBK Blood Lead Model Input Values

Group	Parameter	Parameter Name	Parameter Value IEUBK Default Age Ranges (Years) 0.5 to 1	1 to 2	2 to 3	3 to 4	4 to 5	5 to 6	6 to 7	Basis/Derivation
Inhalation	Daily ventilation rate (cubic meters [m^3]/day)	Ventilation rate	5.4	8.0	9.5	10.9	10.9	10.9	12.4	U.S. EPA Child-Specific Exposure Factors Handbook (2008b) with interpolation for intermediate ages
	Absolute inhalation absorption fraction (unitless)	Lung absorption	0.42							U.S. EPA (1989)
	Indoor air Pb concentration	Indoor air Pb concentration (percentage of outdoor)	100%							These values are taken directly into account when developing the exposure concentrations
	Time spent outdoors	Time spend outdoors (hours/day)	Not used							
Drinking Water Ingestion	Water consumption (L/day)	Water consumption (L/day)	0.36	0.271	0.317	0.349	0.380	0.397	0.414	U.S. EPA Child-Specific Exposure Factors Handbook (2008b) with interpolation for intermediate ages
	Water Pb concentration (µg/L)	Lead concentration in drinking water (µg/L)	4.61							GM of values reported in studies of United States and Canadian populations (residential water) as cited in U.S. EPA (2006), section 3.3 Table 3-10), as in the Lead NAAQS (U.S. EPA, 2007a) and Lead Renovation, Repair, and Painting Rule (U.S. EPA, 2007b)
	Absolute absorption (unitless)	Total percent accessible (IEUBK)	50 % (Single value used across all age ranges)							Assumed similar to dietary absorption (see "Total percent accessible" under Diet below), as in the Lead NAAQS (U.S. EPA, 2007a) and Lead Renovation, Repair, and Painting Rule (U.S. EPA, 2007b)

Table 17. IEUBK Blood Lead Model Input Values

Group	Parameter	Parameter Name	Parameter Value IEUBK Default Age Ranges (Years) 0.5 to 1	1 to 2	2 to 3	3 to 4	4 to 5	5 to 6	6 to 7	Basis/Derivation
Diet	Dietary Pb intake (µg/day)	Dietary Pb intake (µg/day)	3.16	2.6	2.87	2.74	2.61	2.74	2.99	Estimates based on the following: (1) Pb food residue data from U.S. Food and Drug Administration (U.S. FDA) Total Diet Study (USFDA, 2001), and (2) food consumption data from NHANES III (CDC, 1997), as in the Lead NAAQS (U.S. EPA, 2007a) and Lead Renovation, Repair, and Painting Rule (U.S. EPA, 2007b)
Diet	Absolute absorption (unitless)	Total percent accessible	50%							Alexander et al. (1974) and Ziegler et al. (1978) as cited in U.S. EPA (2006, section 4.2.1), as in the Lead NAAQS (U.S. EPA, 2007a) and Lead Renovation, Repair, and Painting Rule (U.S. EPA, 2007b)
Outdoor Soil/Dust and Indoor Dust Ingestion	Outdoor soil/dust and indoor dust weighting factor (unitless)	Outdoor soil/dust and indoor dust ingestion weighting factor (percent outdoor soil/dust)	45%							This is the percent of total ingestion that is outdoor soil/dust. Value reflects best judgment and consideration (results published by van Wijnen et al. (1990), as cited in (U.S. EPA, 1989), as in the Lead NAAQS (U.S. EPA, 2007a) and Lead Renovation, Repair, and Painting Rule (U.S. EPA, 2007b)
Outdoor Soil/Dust and Indoor Dust Ingestion	Total indoor dust + outdoor soil/dust ingestion (mg/day)	Amount of outdoor soil/dust and indoor dust ingested daily (mg)	60	110	110	110	110	110	110	U.S. EPA Child-Specific Exposure Factors Handbook (2008b), excluding cases of soil-pica and geophagy
Outdoor Soil/Dust and Indoor Dust Ingestion	Absolute gastrointestinal absorption (outdoor soil/dust and indoor dust) (unitless)	Total percent accessible (IEUBK)	0.30 for both outdoor soil/dust and indoor dust							Reflects evidence that Pb in indoor dust and outdoor soil/dust is as accessible as dietary Pb and that indoor dust and outdoor soil/dust ingestion may occur away from mealtimes (U.S. EPA 1989), as in the Lead NAAQS (U.S. EPA, 2007a) and Lead Renovation, Repair, and Painting Rule (U.S. EPA, 2007b)

Table 17. IEUBK Blood Lead Model Input Values

Group	Parameter	Parameter Name	Parameter Value IEUBK Default Age Ranges (Years) 0.5 to 1	1 to 2	2 to 3	3 to 4	4 to 5	5 to 6	6 to 7	Basis/Derivation
Other	Maternal PbB (µg/dL)	Maternal PbB concentration at childbirth, µg/dL	0.847							NHANES 2007-2008, national weighted GM of all women aged 18-45 (CDC, 2009)

ALM Parameters

The parameters for the ALM were either set equal to the defaults or were set equal to the values in IEUBK. In particular, the background adult blood lead was set equal to the recommendation of U.S. EPA (2009g) following evaluation of the NHANES 1999-2004 survey data. The parameters are shown in Table 18. The background value may include exposure to indoor dust even though the ALM was adapted to directly apply the biokinetic slope factor to the dust ingestion. However, because the model is linear, the incremental blood lead arising from lead in wheel weights will be unaffected by the choice of the background value.

Table 18. Input Variables and Sources for the Adapted ALM Model

Definition	Variable	Value	Source
Soil + Dust Ingestion Rate, g/day	IR_{S+D}	0.05	U.S EPA (2003), ALM default
Weighting factor; proportion of IR_{S+D} which is soil	W_s	0.45	U.S EPA (1989), same as in IEUBK
Soil and Dust Lead Absorption Fraction	AF_{S+D}	0.12	U.S EPA (2003), ALM default
Biokinetic Slope Factor, µg/dL per µg/day	BKSF	0.4	U.S EPA (2003), ALM default
Background Adult Blood Lead, µg/dL	$PbB_{0,adult}$	1	U.S EPA (2009g), ALM default

4.6 Media Concentrations

First, the AERMOD model was used to estimate air concentrations at each modeled yard. Because these air concentrations are not affected by the soil concentrations (since resuspension is not included, see Section 4.3.1) or housing vintage, scenarios that differ only by these variable definitions will have the same air concentrations. In other words, the urban pre-1940 and post-1980 (Scenarios A and B) have the same air concentrations, as do the downtown rural pre-1940 and post-1980 (Scenarios C and D) scenarios.

Scenarios A and B – Urban pre-1940 (A) and post-1980 (B)

The concentrations in the receptor yards relative to the high and low volume streets for the urban scenario 3 km grid are shown in Figure 13. As discussed in Section 4.2.2, the AERMOD grid represents intersecting streets separated by the typical block length in the proxy city. This proxy city is a Northeastern city with multifamily homes and small yards. High traffic volume streets occur every kilometer with low traffic volume streets between them. In this figure, the light blue lines represent low volume streets, the dark blue lines represent high volume streets, and the colored dots each represent a single yard.

The highest annual-average concentration occurs just to the southeast of the central intersection of the high traffic volume streets and is indicated with a star. At this point, the concentration is 0.017 $\mu g/m^3$, and the total deposition (wet and dry) is 0.0011 g/m^2/year. The modeled concentration can be compared with the total concentration of 0.025 $\mu g/m^3$estimated from the AQS monitors (see section 4.4.2). In initial modeling efforts when street cleaning was not taken into account in the estimation of the lead emission rate, the modeled concentration was 0.054 $\mu g/m^3$, which is above the total concentration. However, the total concentration should include the contribution from wheel weights. This observation indicated the scenario was yielding unrealistically high air concentrations and the cleaning frequency calculation was included to ensure more reasonable modeling results were achieved.

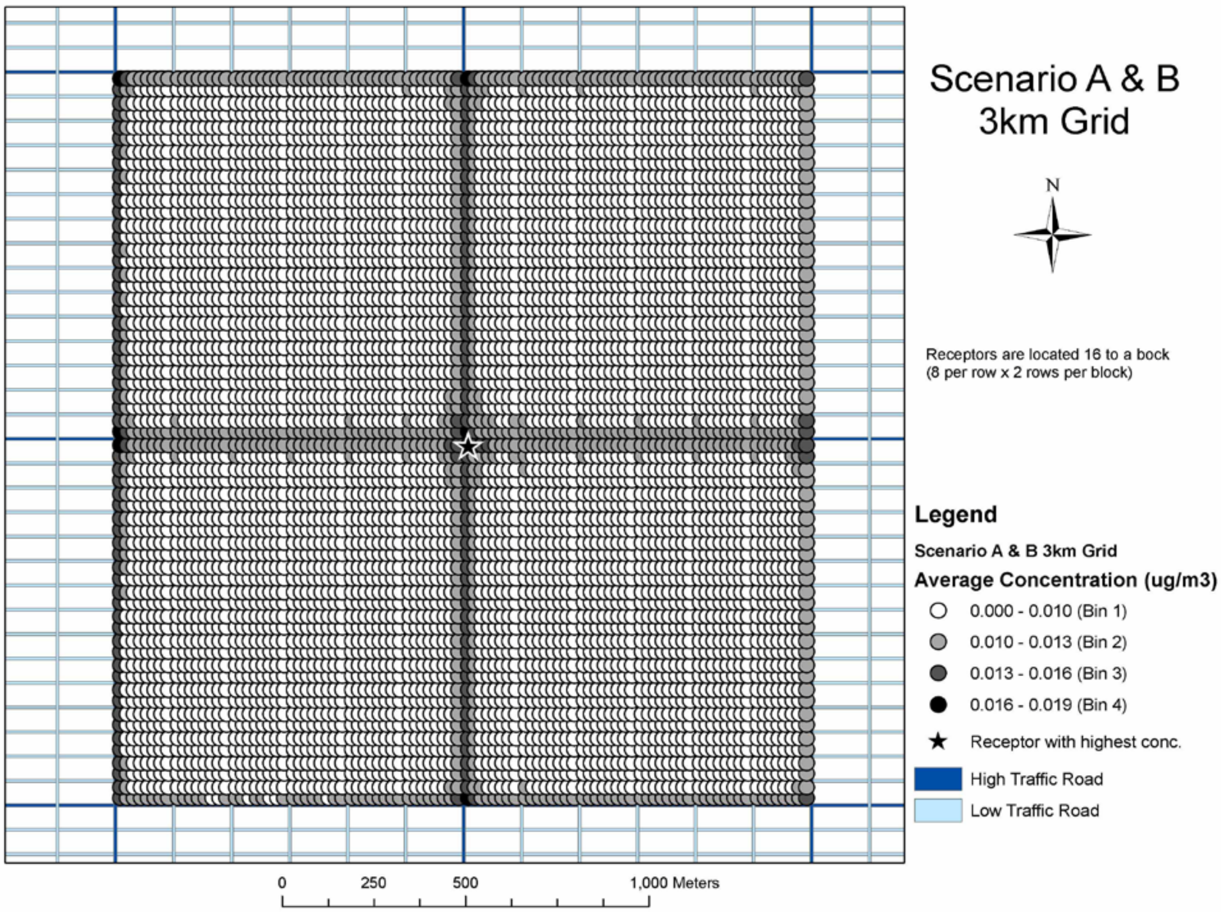

Figure 13. Modeled Concentrations in the Urban Scenario A and B, 3km Grid

Scenarios C and D – Downtown rural, pre-1940 (C) and post-1980 (D)

The concentrations in the receptor yards relative to the high and low volume streets for the rural scenario are shown in Figure 14. The highest annual-average concentration occurs just to the southeast of the central intersection of the high volume traffic. At this point, the concentration is 7.8E-4 $\mu g/m^3$, and the total deposition (wet and dry) is 5.3E-5 g/m^2/year. The modeled concentration can be compared with the total concentration of 0.010 $\mu g/m^3$.

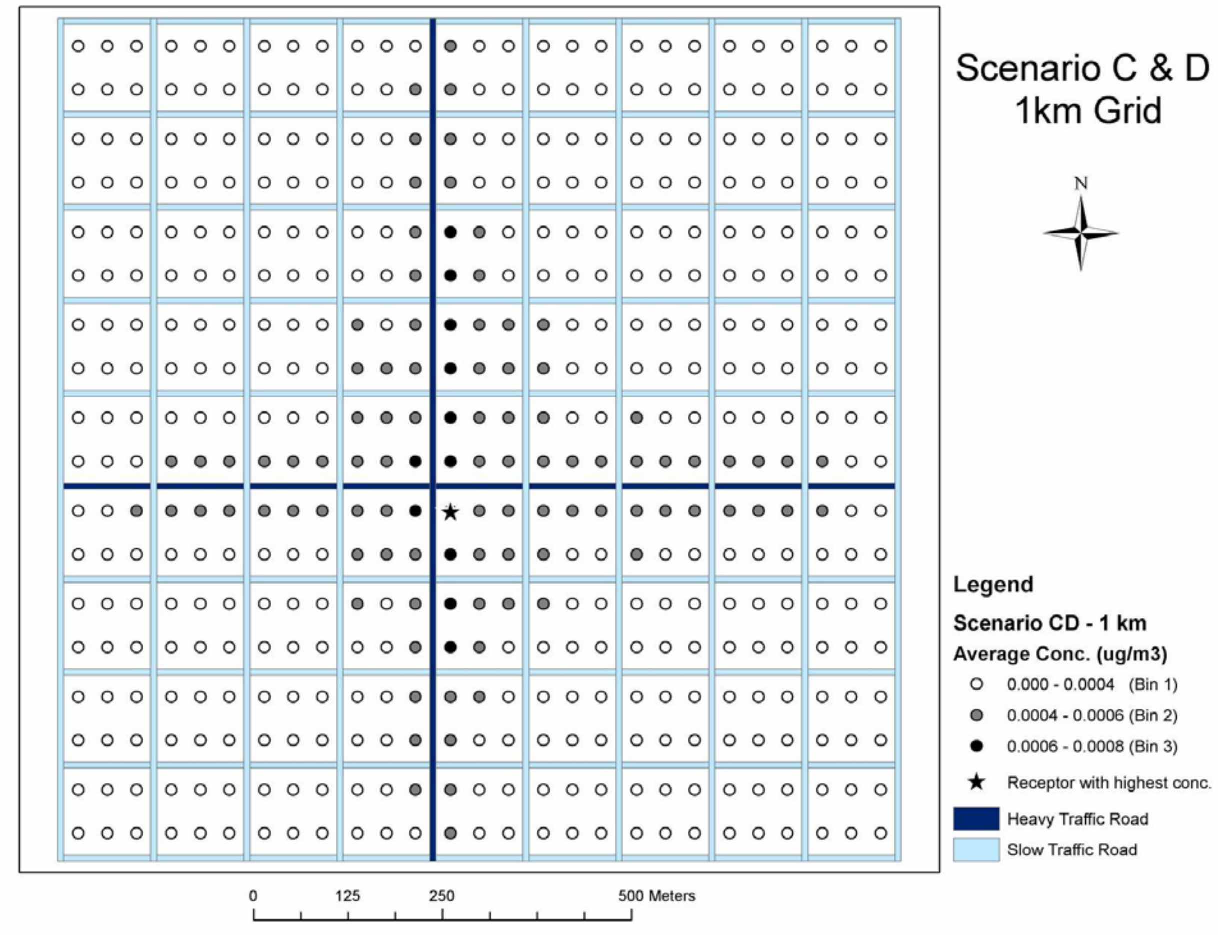

Figure 14. Modeled Concentrations in the Rural Scenario C and D, 1 km Grid

Scenario E – Suburban, post-1980

The concentrations in the receptor yards relative to the high and low volume streets for the suburban scenario 2 km grid are shown in Figure 15. The highest annual-average concentration occurs just to the southeast of the central intersection of the high volume traffic. At this point, the concentration is 2.1E-3 $\mu g/m^3$, and the total deposition (wet and dry) is 1.4E-4 g/m^2/year. The modeled concentration can be compared with the total concentration of 0.014 $\mu g/m^3$.

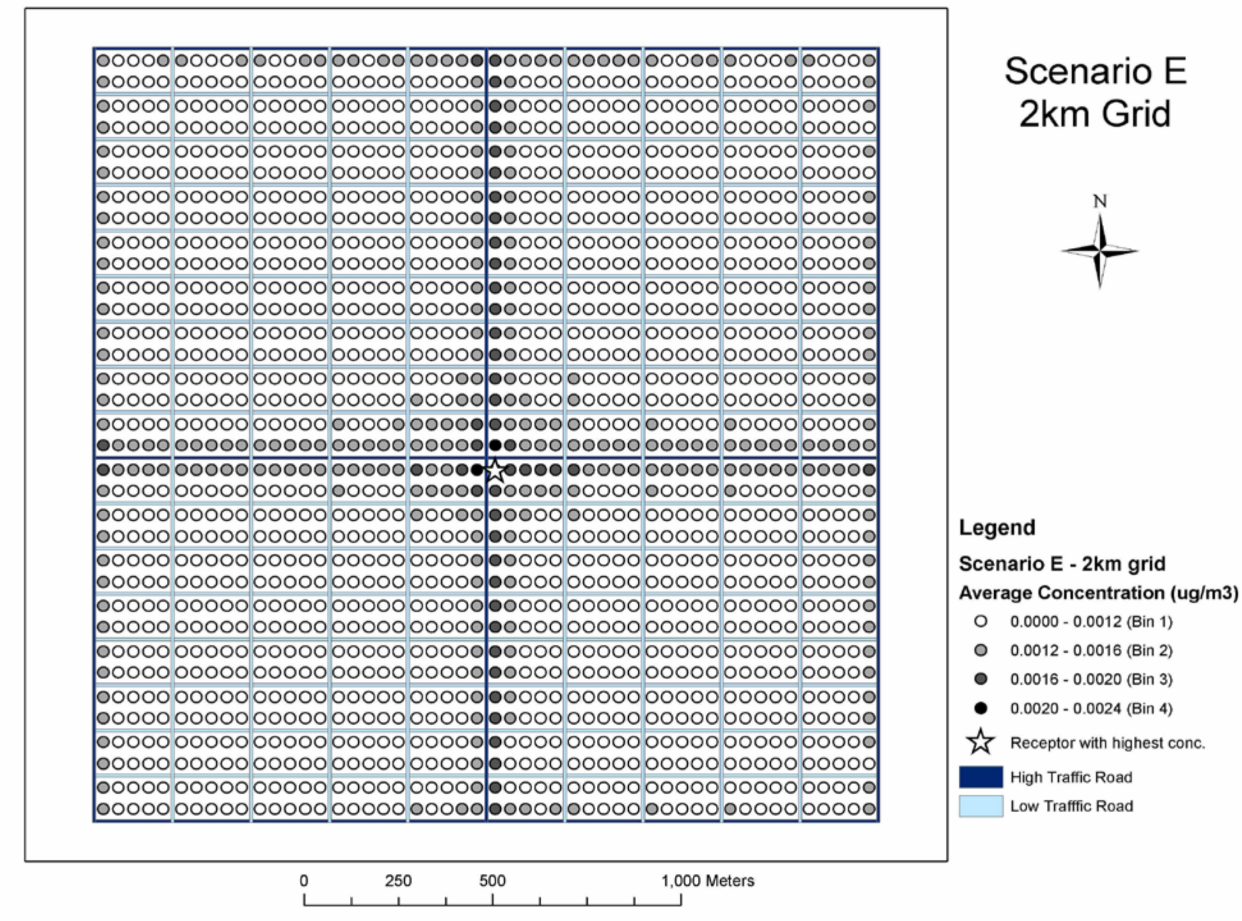

Figure 15. Modeled Concentrations in the Suburban Scenario E, 2 km Grid

Summary of Media Concentrations in the Modeled Scenarios

In each scenario, the modeled air concentrations were binned from lowest (Bin 1) to highest (Bin 3 or 4) concentration intervals that span the range of modeled concentrations in the domain. The bins were selected so that each scenario had three or four bins and the bin boundaries were equally-spaced. Then, the percentage of yards in each concentration bin was calculated using all the modeled yards on the eastern side of the grid. Because the wind is predominantly from the western direction, the eastern side of the grid has a larger contribution from upwind wheel weight emission and thus has a higher level of concentration precision than the western side of the grid. Table 19 shows the bin definitions and the percentage of eastern yards in each bin for the modeled scenarios.

Next, the mean air concentration and deposition was calculated in each bin for each scenario. These concentrations were then used to calculate both the soil and dust concentrations corresponding to these mean concentrations. In addition, the maximum air concentration and deposition in the domain were used to find the media concentrations at the maximally exposed home. Table 20 shows these media concentrations calculated from the AERMOD modeling, the yard soil module, and the indoor dust module. The total media concentration estimates are presumed to include both the wheel weight and other lead source contributions. The wheel weight contribution in Table 20 represents the portion of the total media concentration that is contributed by lead wheel weights. In the case of the dust concentration, this contribution is only approximate since the dust regression equation is nonlinear. The dust concentration was found using the 1) the total soil concentration and 2) the total soil concentration minus the wheel weight contribution and then subtracting 2) from 1). In general, the wheel weight contributions are a small percentage of the total soil and dust concentrations, particularly in the high soil concentration and earlier housing vintage cases. The air concentration contribution is larger, varying from 8% in the rural case up to 70% in the urban case. This large contribution in the urban case is surprising, since resuspended contaminated soil and industrial sources are expected to be the dominant sources in urban environments. However, the total air concentration value itself is highly uncertain, since it is calculated from a network of monitors placed in a variety of locations and which are not necessarily nationally representative. Efforts were made to filter out monitors whose modeling objective was to monitor industrial sources; thus, the total air concentration value may be low for a typical inner-city urban environment, and a true air concentration value is difficult to estimate. However, as will be discussed in the next section, the air concentration does not significantly impact the blood lead estimates; instead, soil and dust intakes are the dominant contributors.

Table 19. Modeled Air Concentration Bin Definitions

Model Scenario	Bin	Maximum Concentration in Bin ($\mu g/m^3$)	Number of Modeled Yards in Bin In Eastern Portion of Domain	Proportion of Modeled Yards in Bin in Eastern Portion of Domain
Scenario A and B Urban	Bin 1	0.0100	2543	85.9%
	Bin 2	0.0130	343	11.6%
	Bin 3	0.0160	70	2.4%
	Bin 4	0.0190	4	0.1%
Scenario C and D Rural	Bin 1	0.0004	207	76.7%
	Bin 2	0.0006	53	19.6%
	Bin 3	0.0008	10	3.7%
Scenario E Suburban	Bin 1	0.0012	674	79.3%
	Bin 2	0.0016	135	15.9%
	Bin 3	0.0020	39	4.6%
	Bin 4	0.0024	2	0.2%

Table 20. Media Concentrations in the Modeled Scenarios

Scenario	Bin	Concentrations					
		Total Air (µg/m³)	Annually-Averaged Wheel Weight Contribution to Air (µg/m³)	Total Soil (µg/g)	Wheel Weight Contribution to Soil (µg/g)	Total Dust (µg/g)	Approximate Wheel Weight Contribution to Dust (µg/g)
Scenario A: Urban area, high soil lead concentration, pre-1940 housing	Bin 1 Mean	0.0250	0.0083	1463.0	25.0	658.5	3.7
	Bin 2 Mean		0.0112		35.0		5.3
	Bin 3 Mean		0.0142		44.8		6.7
	Bin 4 Mean		0.0169		54.7		8.3
	Max		0.0174		55.7		8.4
Scenario B: Urban area, high soil lead concentration, post-1980 housing	Bin 1 Mean	0.0250	0.0083	1463.0	25.0	427.5	2.4
	Bin 2 Mean		0.0112		35.0		3.4
	Bin 3 Mean		0.0142		44.8		4.4
	Bin 4 Mean		0.0169		54.7		5.4
	Max		0.0174		55.7		5.5
Scenario C: Rural area, high soil lead concentration, pre-1940 housing	Bin 1 Mean	0.0100	0.0003	656.0	0.9	504.9	0.2
	Bin 2 Mean		0.0005		1.4		0.4
	Bin 3 Mean		0.0007		2.1		0.5
	Max		0.0008		2.5		0.6
Scenario D: Rural area, low soil lead concentration, post-1980 housing	Bin 1 Mean	0.0100	0.0003	12.0	0.9	87.2	2.2
	Bin 2 Mean		0.0005		1.4		3.6
	Bin 3 Mean		0.0007		2.1		5.4
	Max		0.0008		2.5		6.6
Scenario E: Suburban area, low soil lead concentration, post-1980 housing	Bin 1 Mean	0.0140	0.0010	37.0	2.7	126.6	3.1
	Bin 2 Mean		0.0013		3.8		4.4
	Bin 3 Mean		0.0018		5.4		6.4
	Bin 4 Mean		0.0023		7.0		8.5
	Max		0.0023		7.1		8.7

4.7 Blood Lead Results for Children Age 0 to 7

The bin-mean media concentrations shown in Table 20 were input into the IEUBK blood lead model with the other inputs shown in Table 17. The childhood age 0 to 7 lifetime average blood lead level was calculated for the total exposure case first. Then, the blood lead level was calculated for each modeled scenario and bin by subtracting the wheel weight contribution to each media concentration from the total media concentration. In this way, the blood lead estimates represent situations where wheel weights are present and where wheel weights are not present, respectively.

The uptakes for each of the exposure media are shown in Table 21. Air, soil, and dust routes of exposure are assumed to include lead wheel weight contributions, while water and dietary routes of exposure do not include lead wheel weight contributions. In general, the soil and dust uptakes are the largest contributors to total lead uptake, with diet and water consumption routes playing an intermediate role. The inhalation uptake plays a relatively minor role in the total uptake. In addition, because the precision in IEUBK in the air concentration is 0.01 $\mu g/m^3$, the contribution by the wheel weights to the inhalation uptake is often not resolved in the different scenarios. Thus, the wheel weight contribution to the air concentration makes minimal difference in the blood lead of the child, but the ultimate deposition of this lead in the yard, the incorporation of the lead into yard soil and indoor dust, and the ingestion of this soil and dust by the child are predicted to result in small changes in the blood lead level.

Table 22 shows the blood lead levels for each scenario for the total exposure case and estimates of the contributions from lead wheel weights. In general, the contributions are on the order of 0.01 to 0.1 μg/dL, with the largest blood lead change in the urban high soil case with a value of 0.25 μg/dL. As stated in the introduction to Section 4, each scenario was constructed to represent specific exposure situations for the target populations in an urban, rural, and suburban environment. Thus, the magnitudes of the blood lead predictions should not be compared to national average values or values of a particular percentile in a nationwide survey. Instead, the incremental changes due to lead wheel weights are the key results from the analysis. The magnitude of these incremental changes will vary according to the total exposure media values selected, and the different scenarios were constructed to assist in determining the range of the variation.

Table 21. Uptake Estimates For Children in the Near-Roadway Scenario

Scenario	Bin	Lifetime Average Uptakes *							
		Total Air (μg/day)	Approx. Wheel Weights Air (μg/day)	Total Soil (μg/day)	Approx. Wheel Weights Soil (μg/day)	Total Dust (μg/day)	Approx. Wheel Weights Dust (μg/day)	Total Dietary (μg/day)	Total Water (μg/day)
Scenario A: Urban area, high soil lead concentration, pre-1940 housing	Bin1 Mean	0.12	0.04	20.3	0.35	11.2	0.06	1.4	0.8
	Bin2 Mean		0.08		0.49		0.09		
	Bin3 Mean		0.08		0.62		0.12		
	Bin4 Mean		0.08		0.76		0.14		
	Max		0.08		0.77		0.14		
Scenario B: Urban area, high soil lead concentration, post-1980 housing	Bin1 Mean	0.12	0.04	20.3	0.35	7.3	0.04	1.4	0.8
	Bin2 Mean		0.08		0.49		0.06		
	Bin3 Mean		0.08		0.62		0.07		
	Bin4 Mean		0.08		0.76		0.09		
	Max		0.08		0.77		0.09		
Scenario C: Rural area, high soil lead concentration, pre-1940 housing	Bin1 Mean	0.041	0.00	9.1	0.01	8.6	0.00	1.4	0.8
	Bin2 Mean		0.00		0.02		0.01		
	Bin3 Mean		0.00		0.03		0.01		
	Max		0.00		0.03		0.01		
Scenario D: Rural area, low soil lead concentration, post-1980 housing	Bin1 Mean	0.041	0.00	0.2	0.01	1.5	0.04	1.4	0.8
	Bin2 Mean		0.00		0.02		0.06		
	Bin3 Mean		0.00		0.03		0.09		
	Max		0.00		0.03		0.11		
Scenario E: Suburban area, low soil lead concentration, post-1980 housing	Bin1 Mean	0.041	0.00	0.5	0.04	2.1	0.05	1.4	0.8
	Bin2 Mean		0.00		0.05		0.08		
	Bin3 Mean		0.00		0.07		0.11		
	Bin4 Mean		0.00		0.10		0.14		
	Max		0.00		0.10		0.15		

* Lifetime average indicates average between ages 0 and 7

Table 22. Blood Lead Estimates For Children in the Near-Roadway Scenario From IEUBK

Scenario	Bin	Lifetime Average Blood Lead *	
		Total (μg/dL)	Approx. Wheel Weights Contribution (μg/dL)
Scenario A: Urban area, high soil lead concentration, pre-1940 housing	Bin1 Mean	9.79	0.11
	Bin2 Mean		0.16
	Bin3 Mean		0.20
	Bin4 Mean		0.24
	Max		0.25
Scenario B: Urban area, high soil lead concentration, post-1980 housing	Bin1 Mean	8.86	0.11
	Bin2 Mean		0.16
	Bin3 Mean		0.20
	Bin4 Mean		0.24
	Max		0.24
Scenario C: Rural area, high soil lead concentration, pre-1940 housing	Bin1 Mean	6.28	<0.01
	Bin2 Mean		0.01
	Bin3 Mean		0.01
	Max		0.01
Scenario D: Rural area, low soil lead concentration, post-1980 housing	Bin1 Mean	1.41	0.02
	Bin2 Mean		0.03
	Bin3 Mean		0.04
	Max		0.05
Scenario E: Suburban area, low soil lead concentration, post-1980 housing	Bin1 Mean	1.76	0.03
	Bin2 Mean		0.04
	Bin3 Mean		0.06
	Bin4 Mean		0.08
	Max		0.08

* Lifetime average indicates average between ages 0 and 7

4.8 Blood Lead Results for Adults

Table 23 shows the adult blood lead predictions for total exposure and for the approximate wheel weight contribution. The wheel weight contributions vary between less than 0.01 to 0.07 μg/dL.

Table 23. Blood Lead Estimates For Adults in the Near-Roadway Scenario From the ALM

Scenario	Bin	Total (µg/dL)	Approx. Wheel Weights Contribution (µg/dL)
Scenario A: Urban area, high soil lead concentration, pre-1940 housing	Bin1 Mean	3.45	0.03
	Bin2 Mean		0.04
	Bin3 Mean		0.06
	Bin4 Mean		0.07
	Max		0.07
Scenario B: Urban area, high soil lead concentration, post-1980 housing	Bin1 Mean	3.14	0.03
	Bin2 Mean		0.04
	Bin3 Mean		0.05
	Bin4 Mean		0.07
	Max		0.07
Scenario C: Rural area, high soil lead concentration, pre-1940 housing	Bin1 Mean	2.38	<0.01
	Bin2 Mean		<0.01
	Bin3 Mean		<0.01
	Max		<0.01
Scenario D: Rural area, low soil lead concentration, post-1980 housing	Bin1 Mean	1.13	<0.01
	Bin2 Mean		0.01
	Bin3 Mean		0.01
	Max		0.01
Scenario E: Suburban area, low soil lead concentration, post-1980 housing	Bin1 Mean	1.21	0.01
	Bin2 Mean		0.01
	Bin3 Mean		0.01
	Bin4 Mean		0.02
	Max		0.02

4.9 Uncertainties in the Near-Roadway Exposure Scenario

The approach used to determine the effect of wheel weights on a hypothetical child or adult's blood lead level was designed to be systematic, to use peer-reviewed models and literature wherever possible, and to use approaches and input values similar to those used in other EPA lead analyses. However, the modeled scenarios are subject to numerous uncertainties. The following list highlights some of these uncertainties:

1. **The Root Study and Lead Emission Rates from Degraded Wheel Weights**. The Root (2000) study calculates the rate of lead wheel weight loss from cars and the fraction of roadway wheel weights degraded per day based on year-long sampling on a road in Albuquerque, NM. However, the methodology and conclusions in the study include many uncertainties such as:

    a. The study assumes that steady state conditions and loss rates can be ascertained by looking at the wheel weight stock on the curb only. However, it is likely that degradation mostly occurs on the roadway after the wheel weights fall off the vehicle and before they migrate to the curb. This omission may result in under-prediction of the degradation rates and the amount of lead mass emitted from the roadway.

b. The study assumes that all loss of wheel weights from the curb area occurs due to degradation. However, it is likely that other loss mechanisms are dominant in the curb area including street cleaning collection, collection by hobbyists, runoff into gutters from rain events, and ejection from the curb area into surrounding bushes or near-roadway areas. This assumption will tend to overestimate the amount of lead degradation and release to the air.

2. **The assumption that all degraded lead is emitted to the air**. The modeling approach assumes that all degraded wheel weight lead in the curb is emitted to the air. However, in reality runoff from rain events will remove some of the lead from the roadway before it is emitted. This assumption will tend to overestimate the lead emission rate from the roadway. This assumption is partially examined in the quantitative analysis below.

3. **The dust concentration in the home is not correlated with the ambient air concentration.** The indoor dust concentration in the home was estimated using a regression equation developed from the HUD survey data. However, ambient air measurements were not available in the survey, so indoor dust could not be correlated with the ambient air concentration. In actuality, penetration of ambient air particles into the home and the subsequent settling of particles onto the floor will affect the indoor dust concentration. It is unknown whether this limitation under- or over-predicts the indoor dust concentration.

4. **The use of proxy cities to represent urban, suburban, and rural communities.** Proxy cities were selected for each of the city types according to the availability of media concentrations and traffic data. These cities were used as the basis for the AERMOD grid. However, within these cities there is a wide variety of roadways with varying traffic volumes, and the grids are not uniform. Also, these cities may or may not be representative of the "average" urban, suburban, or rural community with respect to either media concentrations or traffic patterns. Thus, the use of these cities yields an illustrative hypothetical modeling scenario only.

5. **The exclusion of yard soil resuspension.** Resuspension of yard dust does not occur and deposition of roadway dust over the roadway does not occur. These assumptions are necessary to allow "decoupling" of the yard and air modeling compartments. To avoid this assumption, a full multimedia model would have to be used, but these models typically do not handle air dispersion as well as AERMOD.

6. **The application of the blood lead models.** The differences in media concentrations when the lead in wheel weights is excluded are small; the resulting differences in blood lead are also low, with blood lead changes on the order of 0.01 to 0.1 μg/dL. These predictions are close to the precision in the blood lead models and the predictive power of the models in this range is limited.

In addition to the issues noted above, some input variables had a wide range of possible values in the literature. In each case, the selected value was plausible, although no attempt was made to determine the overall probability that the combination of parameter values would exist in a single home or population. In order to determine the effect of these estimates, the child modeling was repeated using alternative values for the six variables deemed of lowest data quality. The six variables correspond to the bold variables in Figure 3.

The first four variables (the wheel weight loss rate, the wheel weight degradation rate, the roadway dust loss rate, and the additional roadway wheel weight removal rate) are all part of the roadway soil module and affect the lead emission rate. All four were either derived from the Root study or were assigned using professional judgment. Thus, the literature could not be used to inform the choice of the value that would be considered an alternative estimate for each variable. As a result, an illustrative case was selected to determine the extent to which the percent change in each variable carried through to the blood lead estimates. The wheel weight loss rate, which was calculated in the Root study, was decreased by a factor of two, resulting in an emission rate which is 50% of the base case emission rate. The loss rate of intact wheel weights due to hobbyist collection, removal into medians, or other processes, which was set at 0% in the base case based on professional judgment, was increased to 50. The loss rate of roadway dust to lateral runoff and other processes, which was set at 0% in the base case based on professional judgment, was increased to 50%. And the degradation rate, which was estimated in the Root study, was changed from 2.7% to 1. The effects of changing these variable values on the emission rate depends on the street cleaning frequency in each of the scenarios.

The other two variables, soil depth and the residence time of lead in soil, are parameters in the residential soil module and the published literature defines a range of values with wide uncertainty and/or variability. For these variables, values resulting in lower wheel weight lead concentrations were selected to explore the effects on the blood lead estimates. In the literature, the soil depth was expected to be between 1 cm and 20 cm, depending on the degree of tilling (yard aeration) and the soil content. For illustrative purposes, the soil depth was changed from the lower point in this range to near the midpoint (10 cm) to determine the effect on the blood lead estimates. The residence times had wide variation in the literature, and 1,000 years was selected for the base. The residence time will depend on the carbon content and other soil properties, and no data were collected in yards. However, many of the newer studies near roadways and in newer (lower carbon content) forests found residence times which had ranges which included 150 years. Thus, this value was selected for the illustrative uncertainty example.

Table 24 shows the change in the blood lead estimates for the different scenarios and compares the base case results with the uncertainty analysis results. For the parameters affecting the emission rate, in each case the change in emission rate carried through the analysis in a nearly linear fashion, such that a 50% decrease in the emission rate resulted in close to a 50% decrease in the blood lead estimates. For the soil variables, changes by a factor of nearly 10 resulted in changes in the blood lead estimates of less than a factor of 10.

Table 24. Estimates of the Blood Lead Changes Resulting from Lead Wheel Weights for the Uncertainty Analysis

Scenario	Bin	WW Blood Lead, Base Case (µg/dL)	WW Loss Rate Decrease by 50%		1% Degradation Rate		Roadway Dust Loss Rate of 50%		Additional Roadway Intact WW Removal Rate of 50%		Soil Depth 10 cm		Soil Residence Time 150 Years	
			WW Blood Lead (µg/dL)	Ratio of Unc. Case and Base Case	WW Blood Lead (µg/dL)	Ratio of Unc. Case and Base Case	WW Blood Lead (µg/dL)	Ratio of Unc. Case and Base Case	WW Blood Lead (µg/dL)	Ratio of Unc. Case and Base Case	WW Blood Lead (µg/dL)	Ratio of Unc. Case and Base Case	WW Blood Lead (µg/dL)	Ratio of Unc. Case and Base Case
Scenario A: Urban area, high soil lead concentration, pre-1940 housing	Bin 1 Mean	0.11	0.05	**0.5**	0.05	**0.4**	0.05	**0.5**	0.02	**0.2**	0.02	**0.2**	0.03	**0.2**
	Bin 2 Mean	0.16	0.07	**0.5**	0.06	**0.4**	0.07	**0.5**	0.02	**0.2**	0.03	**0.2**	0.03	**0.2**
	Bin 3 Mean	0.20	0.10	**0.5**	0.08	**0.4**	0.10	**0.5**	0.03	**0.2**	0.04	**0.2**	0.04	**0.2**
	Bin 4 Mean	0.24	0.12	**0.5**	0.10	**0.4**	0.12	**0.5**	0.04	**0.2**	0.04	**0.2**	0.05	**0.2**
	Maximum	0.25	0.12	**0.5**	0.10	**0.4**	0.12	**0.5**	0.04	**0.2**	0.04	**0.2**	0.06	**0.2**
Scenario B: Urban area, high soil lead concentration, post-1980 housing	Bin 1 Mean	0.11	0.05	**0.5**	0.05	**0.4**	0.05	**0.5**	0.02	**0.2**	0.02	**0.2**	0.03	**0.2**
	Bin 2 Mean	0.16	0.07	**0.5**	0.06	**0.4**	0.07	**0.5**	0.02	**0.2**	0.03	**0.2**	0.04	**0.2**
	Bin 3 Mean	0.20	0.09	**0.5**	0.08	**0.4**	0.09	**0.5**	0.03	**0.2**	0.04	**0.2**	0.04	**0.2**
	Bin 4 Mean	0.24	0.11	**0.5**	0.10	**0.4**	0.11	**0.5**	0.04	**0.2**	0.04	**0.2**	0.05	**0.2**
	Maximum	0.24	0.12	**0.5**	0.10	**0.4**	0.12	**0.5**	0.04	**0.2**	0.04	**0.2**	0.05	**0.2**
Scenario C: Rural area, high soil lead concentration, pre-1940 housing	Bin 1 Mean	<0.1	<0.1	--	<0.1	--	<0.1	--	<0.1	--	<0.1	--	<0.1	--
	Bin 2 Mean	0.01	<0.1	--	<0.1	--	<0.1	--	<0.1	--	<0.1	--	<0.1	--
	Bin 3 Mean	0.01	<0.1	--	<0.1	--	<0.1	--	<0.1	--	<0.1	--	<0.1	--
	Maximum	0.01	0.01	**0.5**	<0.1	--	0.01	**0.5**	<0.1	--	<0.1	--	<0.1	--
Scenario D: Rural area, low soil lead concentration, post-1980 housing	Bin 1 Mean	0.02	0.01	**0.5**	0.01	**0.6**	0.01	**0.5**	<0.1	--	<0.1	--	<0.1	--
	Bin 2 Mean	0.03	0.01	**0.5**	0.02	**0.6**	0.01	**0.5**	<0.1	--	<0.1	--	<0.1	--
	Bin 3 Mean	0.04	0.02	**0.5**	0.03	**0.6**	0.02	**0.5**	<0.1	--	0.01	**0.1**	0.01	**0.2**
	Maximum	0.05	0.02	**0.5**	0.03	**0.6**	0.02	**0.5**	<0.1	--	0.01	**0.1**	0.01	**0.2**
Scenario E: Suburban area, low soil lead concentration, post-1980 housing	Bin 1 Mean	0.03	0.01	**0.5**	0.01	**0.5**	0.01	**0.5**	<0.1	--	0.00	**0.1**	0.01	**0.2**
	Bin 2 Mean	0.04	0.02	**0.5**	0.02	**0.5**	0.02	**0.5**	<0.1	--	0.01	**0.1**	0.01	**0.2**
	Bin 3 Mean	0.06	0.03	**0.5**	0.03	**0.5**	0.03	**0.5**	0.01	**0.1**	0.01	**0.1**	0.01	**0.2**
	Bin 4 Mean	0.08	0.04	**0.5**	0.04	**0.5**	0.04	**0.5**	0.01	**0.1**	0.01	**0.1**	0.01	**0.2**
	Maximum	0.08	0.04	**0.5**	0.04	**0.5**	0.04	**0.5**	0.01	**0.1**	0.01	**0.1**	0.01	**0.2**

Unc. = Uncertainty; WW = lead wheel weights; Ratios are not calculated for blood lead values below 0.01 µg/dL.

5. HOME MELTING EXPOSURE SCENARIO

In addition to the exposure pathway described previously, wheel weights that are lost from cars or removed by tire shop employees can also be collected by home hobbyists, who melt the wheel weights and produce a variety of hobby related items including lead fishing lures and sinkers, lead soldiers, and bullets. A case of acute lead poisoning was reported by the State of Alaska (State of Alaska Department of Health and Social Services, 2001) when a man turned his home hobby of fish sinker and ingot casting into a cottage industry and moved indoors into poorly ventilated space. This case indicates that exposure potential from inhalation of fumes and ingestion of indoor contaminated dust exists and can be quite high. Thus, this exposure scenario estimates the inhalation exposure concentration and garage dust loading for a child and adult present during a single melting event. Section 5.1 discusses the selected assessment method, Section 5.2 discusses the parameter selection, Section 5.3 presents the exposure concentrations and loadings, and Section 5.4 discusses the dominant uncertainties.

5. 1 Assessment Method Selected

Home melting of wheel weights can occur outside or inside the home. In this approach, the wheel weights are assumed to be melted in a garage, during a one hour session, with both an adult and child present. The child is included in the scenario to account for exposure to the more sensitive population, although the plausibility of a child being present during the event is unknown. Melting is usually achieved through the use of an electric pot, sold for this purpose, or with a propane burner. When either of these heating methods is employed, the lead, upon melting, will maintain equilibrium with the air above the pot. The air pressure will be equal to the saturation vapor pressure at the temperature of the interface of the lead and air. The heat generated will also cause a buoyant plume to form, as the heated air with lead vapor and combustion by-products directly above the pot will be less dense than the surrounding air. Agitation of the pot by stirring or mixing, which lowers the surface tension of the molten lead, will also release lead vapor. Modeling the processes of emission would be computationally rigorous, requiring computation fluid dynamic (CFD) modeling in connection with a mass balance model of the garage. However, this approach would involve numerous parameters and each would contribute to the overall uncertainty. Given the uncertainty in the emission rate from the pot, a simpler approach was selected.

Such a simpler, high-end method is the use of the saturation vapor pressure approach. The saturation vapor pressure approach assumes that the concentration of lead in indoor air throughout the room during the melting operation is equal to the equilibrium vapor pressure of lead at its melting temperature. This approach was recommended by Gurumurthy (2005) for modeling occupational exposures to chemicals when the data on workplace dimensions and practices are not available or are highly uncertain. Although this approach is high-end, it is likely to approximate the airborne lead concentration directly over the pot, which is where the home hobbyist would most likely be located during the melting event while stirring the metal and pouring it into casts.

The saturation vapor pressure approach assumes that the concentration of airborne lead is equal to the equilibrium vapor pressure at the temperature of melting for the entire duration of the melting event. In a closed system, lead vapor is formed above the surface of the molten lead and ultimately attains a thermodynamic equilibrium if the system is not perturbed. When such equilibrium is established, the concentration of lead expressed in pressure units is equal to the vapor pressure of lead at the temperature of the liquid. In reality, such equilibrium, if reached, will take time and will not be instantaneous. In addition, the concentration above the pot will be diluted in the remainder of the room and it will take time for the concentration in the whole garage to match the saturation vapor pressure (if it ever does). For this reason, the saturation vapor pressure represents an upper bound on the exposure air concentration and is appropriate for a high-end analysis. This approach does not rely on knowing the amount of lead in the pot or the size of the pot. Instead, by assuming the concentration in the garage instantaneously equals the saturation vapor pressure, only the chemical properties of lead, the temperature at which the lead is melted, and the duration of the melting event are needed. The two melting temperatures are related to their respective saturation vapor pressures by the Clausius-Clapeyron Equation (Schwarzenbach, 2003).

$$P_2 = P_1 \exp\left(\frac{-H_{vap}}{R}\right)\left(\frac{1}{T_2} - \frac{1}{T_1}\right)$$

where:

P_2 = saturation vapor pressure at the temperature of melting (Pa)

P_1 = saturation vapor pressure at the reference temperature (Pa)

H_{vap} = enthalpy of vaporization of lead, 179 kJ/mol

R = Universal gas law constant (8.314 J K^{-1} mol^{-1})

T_2 = melting temperature (K)

T_1 = reference temperature (K)

The reference saturation vapor pressure and temperature, 1.33 kPa and 1433K, respectively were taken from the Toxnet Hazardous Substances Database information for lead (USNLM, 2010).

The calculated saturation vapor pressures associated with each melting temperature were converted to airborne concentrations by the ideal gas law, presented below.

$$Conc = \frac{P * MW}{RT}$$

where:

Conc = Concentration (μg/m^3)

P = pressure (Pa)

MW = molecular weight of lead (207.4 g/mol)

R = Universal gas law constant (8.314 m^3 Pa K^{-1} mol^{-1})

T = temperature (K)

The airborne concentrations were then related to the inhalation dose of lead for each scenario during the melting duration. The dose was estimated to be equal to the product of the airborne concentration of lead multiplied by the inhalation rate multiplied by the inhalation absorption fraction multiplied by the time of exposure.

During and following melting, the lead vapor in the air will cool and form particles that will settle and mix with the garage dust. By assigning a standard volume (both in height and area) to the garage and an appropriate particle deposition rate and air exchange rate for the garage, it is possible to estimate the floor lead dust loading in the garage. A simple mass-balance model was constructed to simulate the dust levels.

For the mass balance model, the changes in the airborne and deposited lead masses in the garage are related by the following two equations:

$$Mass_{air}(t=0) = Conc \times A \times h$$

$$Mass_{floor}(t=0) = D \times Mass_{air}(t=0)$$

$$\frac{dMass_{air}}{dt} = Mass_{air}\left(-\lambda - D\right)$$

and

$$\frac{dMass_{floor}}{dt} = Mass_{air} \times D$$

where:

$Mass_{air}$ = mass of lead deposited in the air of the garage, μg

$Mass_{floor}$ = mass of lead deposited on the floor of the garage, μg

$Mass_{air}(t=0)$ = the initial condition for the air mass just following the melting event

$Mass_{floor}(t=0)$ = the initial condition for the floor mass just following the melting event

t = time (hrs)

λ = typical garage air exchange rate

D = particle loss coefficient for deposition

A = the floor area of the garage

h = the height of the garage

This approach separates the exposure time into the time during the melting event and the time after the event. During the melting event, the air concentration is set equal to the saturation vapor pressure. The air exchange cannot be incorporated into this portion of the exposure event, since the vapor pressure model is a purely physicochemical model and does not allow for air exchange and other physical processes to mediate the concentration. Thus, it is assumed during this time that the floor loading at the end of the melting event is represented by the deposition rate multiplied by the total mass in the air and multiplied by the duration of the melting event (one hour).

After the melting event, the above equations are numerically integrated until the air concentrations reach only 0.1% of their during-melting levels. The resulting floor mass is divided by the floor area to estimate the floor lead loading at that time. It is worthwhile to note that because the concentration is assumed to be constant throughout the volume of the garage and the floor loading is normalized on an area basis, the floor loading for any size of garage would only vary with differences in ceiling height and would not change based on the footprint (square footage) of the garage.

Because this scenario represents high-end exposure to a single melting event, blood lead levels were not calculated. Blood lead is a long-term measure of exposure and the models are less suited to very short acute exposures, such as a single melting event. Combining melting events into a longer exposure profile would involve knowing the frequency of events and the cleaning frequency and efficiency in the garage, all of which are expected to be highly variable and are uncertain in the literature. Thus, the inhalation exposure concentrations and single-event dust loadings are the exposure metrics in this analysis, and blood lead levels were not estimated.

5.2 Parameter Selection

Representative Temperatures for Lead During Melting

Several sources provide recommendations to home bullet casters on the temperature ranges to maintain throughout the melting process. The how-to article “Bullet Casting for Beginners” (Boothroyd, undated) mentions that many casters melt lead at a temperature near 800^{o}F (427^{o}C). Because the wheel weights used in the melting contain a lead-antimony-tin alloy, the melting temperature is below the melting temperature of pure lead and is near 565^{o}F (296^{o}C, Boothroyd, undated). The author recommends keeping the temperature near this melting point at a temperature of 650^{o}F (343^{o}C). This will ensure the lead has melted but will not be hot enough to create “whiskers” as lead seeps through cracks in the mold when poured. Such whiskers will affect the performance of the bullet once used. In addition, keeping the temperature lower allows the caster to make many more bullets in a fixed amount of time because the cooling time is shorter. Thus, casters have an incentive to keep the melting temperature lower.

Because the performance of the bullet will be a priority of the caster, typical temperatures for bullet casting are likely between 650^{o}F (343^{o}C) and 800^{o}F (427^{o}C). Thus, these two temperatures are selected for the analysis. Although the pots can melt at temperatures up to 1000 ^{o}F (538^{o}C, Boothroyd, undated), smoke would be created and would induce the caster to turn down the heat. In addition, casting at such a higher temperature will lead to impurities in the bullet. Thus, this temperature is included as an upper bound, but casters are not expected to maintain the pot at this temperature for long periods of time.

Duration of the Melting Event

The how-to article “Bullet Casting for Beginners” (Boothroyd, undated) mentions that casters can make up to 330 bullets in a single hour if the temperatures are kept near 650^{o}F (343^{o}C). Melting at higher temperatures yields approximately a 1/3 reduction in efficiency, resulting in approximately 100 bullets per hour. It is expected that this hour-long yield will be sufficient for a hobbyist, so a duration of one hour was selected as the melting duration.

Breathing Rates and Absorption Fractions

The average inhalation rate for children aged 2 to 16 during moderate activity is 1.37 m^3/hr (U.S. EPA, 2008b) and the adult inhalation rate during moderate activity is 1.6 m^3/hr (U.S. EPA, 1997). The absorption fraction was set equal to 0.42 as used in IEUBK (see Section 3.5.2).

Garage Height

The garage was assumed to have a height of 10 ft (or 3.0 m). This value is based on professional judgment.

Garage Air Exchange Rate

The air exchange rate (AER) for an attached residential garage was set at 1.24 hr^{-1} following EPA’s exposure modeling guidance (Johnson, 2002.)

Particle Deposition Rate

The particle loss coefficient, D, is correlated with particle size. Nazaroff (2004) reports different deposition rates for different size particles, with deposition rates ranging from 0.1 to 2 hr^{-1}. A review of the literature suggests that a mass median aerodynamic particle diameter of 5 μm can be expected for lead melting operations (Donguk and Namwon, 2004). This size corresponds to a deposition rate of 2 hr^{-1}. Because the mass deposited on the floor is linearly correlated with the loss coefficient, alternative particle diameters of 0.01, 0.1, and 1 μm were also analyzed (corresponding to deposition rates of 1 hr^{-1}, 0.1 hr^{-1}, and 1 hr^{-1}, respectively).

5.3 Media concentration and inhalation results

The intermediaries and results of this approach are presented in Table 25 below, with inputs in normal font and results in bold font. As shown, the dose for both adults and children is much higher (nearly two orders of magnitude) at the middle temperature versus the lower temperature. Adult doses are slightly higher than child doses due to the higher inhalation rate of adults.

Table 25. Summary of Model Intermediaries and Results for the Home Melting Scenario

Variable Description	Units	Melting at 650°F	Melting at 800°F	Melting at 1000°F
Melting temperature	°C	343	427	538
Saturation vapor pressure	kPa	2.90E-09	1.87E-07	1.28E-05
Airborne Lead Concentration	**$\mu g/m^3$**	**0.24**	**15.7**	**1070**
Adult Inhalation Exposure per hour	**μg/hour**	**0.163**	**10.5**	**719**
Child Inhalation Exposure per hour	**μg/hour**	**0.139**	**9.0**	**614**
Garage floor dust loading, d_p = 5 μm	**$\mu g/ft^2$**	**0.177**	**11.4**	**781**
Garage floor dust loading, d_p = 0.01 μm or 1 μm	**$\mu g/ft^2$**	**0.0978**	**6.31**	**431**
Garage floor dust loading, d_p = 0.1 μm	**$\mu g/ft^2$**	**0.0118**	**0.76**	**52.1**

The results were compared to monitoring data collected by OSHA at facilities that manufactured sporting goods, such as fishing tackle and bullets and were likely to include casting of lead (OSHA, 2010). Data were available from a total of 62 facilities where 394 personal 8-hr samples were collected. It is assumed that personal samples, collected in the breathing zone of the worker most closely approximate the concentrations calculated with the saturation vapor pressure approach and the concentration to which the home hobbyist would be exposed. Of the samples collected, 297 were below the detection limit for lead. It was assumed that lead casting did not occur at these facilities. Of the personal samples that were above detection, the median lead concentration for personal samples was 32.4 $\mu g/m^3$ and the mean lead concentration was 172 $\mu g/m^3$, with a range from 3.3 to 4,800 $\mu g/m^3$. Personal sampling concentrations can be affected by many things, including the size of the lead melting source, the proximity of the worker to the source, and building characteristics including the building ventilation system. The agreement between the OSHA collected personal sampling concentrations and the airborne lead concentrations calculated using the saturation vapor pressure approach suggests that the results are feasible and appropriate.

5.4 Uncertainties in the Home Melting Scenario

The modeling of the home melting scenario uses a highly simplified approach that is very sensitive to the melting temperature. The approach assumes the airborne lead concentration at the site of the melting (the garage) is equal to the saturation vapor pressure throughout the time spent melting lead and does not allow for any gradual achievement of steady state. In addition, this approach does not consider removal mechanisms, such as removal by deposition or ventilation.

6. IQ DECREMENTS FOR THE NEAR-ROADWAY SCENARIO

In order to facilitate the cost-benefit analysis in support of the wheel weights rule, IQ decrements were calculated for children exposed to lead wheel weights in the Near-Roadway Scenario. Section 6.1 presents the approach used for the modeling and Section 6.2 presents the results.

6.1 IQ Estimation Approach

For children, the human health endpoint selected was IQ decrement. The IQ module estimates the IQ decrement associated with the lifetime-average blood lead value between age zero and seven as depicted in Figure 16.

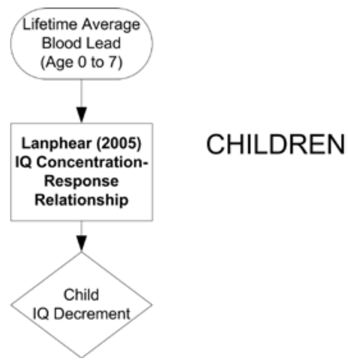

Figure 16. Flowchart Showing the Approach for the IQ Decrement Module

The Lanphear pooled analysis looked at the IQ decrements in children as a function of their lead exposure in the pooled data from seven studies and 1,333 children. The concentration-response functions from this paper were used in the exposure analysis for the review of the Lead NAAQS (U.S. EPA, 2007a) and for the LRRP rule (U.S. EPA, 2007b) and represent the functions based on the largest number of subjects and across the widest exposure range in the literature. Thus, these concentration-response functions were also selected for this analysis. For adults, the data in the literature remain inconclusive as to the most sensitive human health endpoint; thus, the adult exposure calculations estimate the blood lead levels only.

IQ decrements were estimated for children age 0 to 7. Lanphear et al. (2005) derived regression relationships between several blood lead metrics (lifetime averages and measurements made concurrently with the IQ test administration) and IQ test results based on linear, cubic spline, log-linear, and piecewise linear equations. The regression using piecewise linear equations and the lifetime blood lead average was selected to analyze the lead wheel weights IQ changes. The model has a blood lead "cutpoint" at 10 μg/dL where the slope of the concentration-response curve goes from a steeper slope at low blood lead levels to a less steep slope at higher blood lead levels. The equation relating blood lead to the change in IQ is then:

$PbB < 1$ $\quad$ $IQ\ change = 0$

$PbB = 1 \text{ to } 10$ $\quad$ $IQ\ change = PbB * -0.88$

$PbB > 10$ $\quad$ $IQ\ change = -8.8 + (PbB - 10) * -0.10$

where:

PbB = Lifetime average of the blood lead level

As shown in the above equations, no IQ changes are predicted for blood lead concentrations less than 1.0 µg/dL. This assumption was made in recognition of the lack of data in this blood lead range in the Lanphear et al. (2005) study cohorts.

6.2 IQ Results

These lifetime blood lead estimates were then input into the IQ concentration-response function to estimate the IQ decrement for each near-roadway exposure scenario. Then, the change in IQ decrement caused by the presence of lead in wheel weights was estimated by subtracting the no wheel weights case from the total exposure case for each scenario and bin. The IQ decrements are shown in Table 26.

Table 26. IQ Decrements for Children in the Near-Roadway Scenario

Scenario	Bin	IQ Decrement	
		Total (IQ Points)	Approx. Wheel Weights Contribution (IQ Points)
Scenario A: Urban area, high soil lead concentration, pre-1940 housing	Bin1 Mean	-8.62	-0.10
	Bin2 Mean		-0.14
	Bin3 Mean		-0.18
	Bin4 Mean		-0.21
	Max		-0.22
Scenario B: Urban area, high soil lead concentration, post-1980 housing	Bin1 Mean	-7.79	-0.10
	Bin2 Mean		-0.14
	Bin3 Mean		-0.17
	Bin4 Mean		-0.21
	Max		-0.21
Scenario C: Rural area, high soil lead concentration, pre-1940 housing	Bin1 Mean	-5.53	> -0.01
	Bin2 Mean		-0.01
	Bin3 Mean		-0.01
	Max		-0.01
Scenario D: Rural area, low soil lead concentration, post-1980 housing	Bin1 Mean	-1.24	-0.01
	Bin2 Mean		-0.02
	Bin3 Mean		-0.04
	Max		-0.04
Scenario E: Suburban area, low soil lead concentration, post-1980 housing	Bin1 Mean	-1.55	-0.03
	Bin2 Mean		-0.04
	Bin3 Mean		-0.05
	Bin4 Mean		-0.07
	Max		-0.07

The change in the IQ decrement due to the presence of the wheel weights is below one IQ point, with maximal changes varying between 0.01 IQ points in Scenario C to 0.2 IQ points in Scenario

A. In general, the wheel weights make a larger percent difference in the rural and suburban cases where exposures are lower and the relative contribution from wheel weights is larger. However, the absolute magnitude of the change in IQ in these scenarios is small. Wheel weights tend to have the largest percent difference in the inhalation exposure route, but because this route produces the smallest total uptakes, the overall contribution of wheel weights is lower than the air concentrations alone might suggest.

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APPENDIX. LEGGETT ADULT BLOOD LEAD PREDICTIONS FOR THE NEAR-ROADWAY SCENARIO

The ALM model was used as the primary method to estimate the adult blood lead contributions from lead wheel weights in the near roadway scenario. An alternative model, the Leggett model, can be applied to adults but tends to predict blood lead values which are higher than those observed in surveys such as the NHANES (U.S. EPA, 2007). However, for comparison purposes, blood lead values estimated from Leggett are included in this appendix.

Leggett Parameters

The Leggett blood lead model was used for the adult scenarios because the IEUBK model only models exposures up to age 7. The Leggett model was run beginning from birth and extending to age 75 with constant media concentrations throughout this lifetime. Unlike the IEUBK model, the Leggett model requires inputs of total inhalation intake and total ingestion intake over the age range modeled. These intakes were calculated the same way as in the IEUBK model in order to be consistent between the two methods. Thus, the total ingestion intake includes intakes from soil, dust, dietary, and water sources. Leggett intake parameters for the age range 0 to 7 years and for all parameters that do not vary by age were taken from the values used in the IEUBK model (see Table 17). Table A-1 shows the age-specific inputs for ages above age 7. This age range was split into two different segments: 7 to 18 (the remainder of childhood) and 18-75 (adulthood). Parameters were taken from the child-specific or general exposure factors handbook where available.

Table A-1. Leggett Blood Lead Model Input Values

Group	Parameter	Parameter Name	Parameter Value		Basis/Derivation
			7 to 18	18 - 75	
Inhalation	Daily ventilation rate (cubic meters [m^3]/day)	Ventilation rate	14.4	13.3	U.S. EPA Child-Specific Exposure Factors Handbook (2008b) with interpolation for intermediate ages; U.S. EPA Exposure Factors Handbook (1997), average of males and females
Drinking Water Ingestion	Water consumption (L/day)	Water consumption (L/day)	0.571	1.47	U.S. EPA Child-Specific Exposure Factors Handbook (2008b) with interpolation for intermediate ages; U.S. EPA Exposure Factors Handbook (1997), average of males and females
Diet	Dietary Pb intake (µg/day)	Dietary Pb intake (µg/day)	3.5	3.5	Based on dietary intake values from the Lead ISA
Outdoor Soil/Dust and Indoor Dust Ingestion	Total indoor dust + outdoor soil/dust ingestion (mg/day)	Amount of outdoor soil/dust and indoor dust ingested daily (mg)	110	50	Child Estimates based on U.S. EPA Child-Specific Exposure Factors Handbook (2008b), excluding cases of soil-pica and geophagy; Adult estimates from U.S. EPA Exposure Factors Handbook (1997).

Leggett Blood Lead Predictions

The adult blood lead concentrations were estimated using the same media concentrations presented in Table 20. These concentrations were input into the Leggett model, and the average blood lead levels during the child-bearing years (assumed to be age 18-45) and during the adult years (assumed to be age 18-75) were calculated. These values are presented in Table A-2.

Table A-2. Blood Lead Values for Adults in the Near-Roadway Scenario from the Leggett Model

Scenario	Bin	Child Bearing Years Average Blood Lead *		Age 18-75 Average Blood Lead	
		Total (µg/dL)	Approx. Wheel Weights Contribution (µg/dL)	Total (µg/dL)	Approx. Wheel Weights Contribution (µg/dL)
Scenario A: Urban area, high soil lead concentration, pre-1940 housing	Bin1 Mean	18.78	0.21	18.77	0.21
	Bin2 Mean		0.29		0.29
	Bin3 Mean		0.37		0.37
	Bin4 Mean		0.45		0.45
	Max		0.46		0.46
Scenario B: Urban area, high soil lead concentration, post-1980 housing	Bin1 Mean	16.28	0.19	16.28	0.19
	Bin2 Mean		0.27		0.27
	Bin3 Mean		0.35		0.35
	Bin4 Mean		0.42		0.42
	Max		0.43		0.43
Scenario C: Rural area, high soil lead concentration, pre-1940 housing	Bin1 Mean	12.74	0.01	12.78	0.01
	Bin2 Mean		0.01		0.01
	Bin3 Mean		0.02		0.02
	Max		0.02		0.02
Scenario D: Rural area, low soil lead concentration, post-1980 housing	Bin1 Mean	4.72	0.03	4.85	0.03
	Bin2 Mean		0.05		0.05
	Bin3 Mean		0.07		0.07
	Max		0.09		0.09
Scenario E: Suburban area, low soil lead concentration, post-1980 housing	Bin1 Mean	5.30	0.05	5.42	0.05
	Bin2 Mean		0.07		0.07
	Bin3 Mean		0.11		0.11
	Bin4 Mean		0.14		0.14
	Max		0.14		0.14

* Child bearing years indicates average between ages 18 and 45